Considerations for addressing indirect land use change in Danish biofuel regulation

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Cerulogy

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Disclaimer

Summary

Renewable energy policy is an important part of the European Union's climate package, and since the first Renewable Energy Directive (RED) was adopted in 2009 support for biofuels has been an important part of EU renewable energy policy. Because support for biofuel supply is intended to help Europe deliver its climate goals, considerable attention has been paid to developing the lifecycle analysis (LCA) of the GHG emissions associated with biofuels production. Lifecyle analysis is an analytical tool that can be used to assess the overall GHG emissions from the GHG sources and sinks that we consider to be associated with the use of a given biofuel.

While the premise of LCA may seem simple enough, complexity is immediately introduced when we stop to ask what it means for a given emissions to be 'associated' with a given biofuel pathway. We find that the emissions recorded for a given fuel can vary significantly depending on how the scope for a LCA is set. In particular, there are two different families of LCA question that we could ask about biofuels, and it turns out that asking different questions can lead to quite different answers.

On the one hand, we have the type of questions that we might want to ask when we are considering whether biofuel mandates represent good climate change policy. These are questions about the consequences of biofuel use, for instance we could ask, "What is the expected change in net global emissions if we require the supply to the transport of an additional unit of biofuel?" When a LCA is used to answer this type of question, we call it a consequential LCA. If the answer to this question is that increasing biofuel supply delivers significant GHG benefits compared to the cost of the policy, then we would conclude that biofuel mandates are a good climate change policy tool. If instead the answer was that mandating biofuels was not expected to deliver net emissions savings, then we would conclude that biofuel mandates were not a good climate policy tool.

While these consequential questions are clearly very relevant, they can also be difficult to answer precisely. Consequential LCA requires modelling the way that the consequences of a policy decision ripple out through the economy. A different type of LCA question, which can be more precisely answered, would be a question like, "what emissions are associated with the processes required to produce a unit of biofuel by growing a given feedstock?" We call an answer to this sort of question an attributional LCA, because it involves deciding which processes in the world can be attributed to the production of a given batch of biofuels and creating an inventory of the associated emissions. Attributional LCA is a simpler task than consequential LCA because it has a much narrower scope. Where a consequential approach might require us to consider changes across the whole agricultural economy, attributional analysis allows us to assess a specific farm. Attributional analysis has great utility as a way to assess the relative efficiency of different processes, and has become a standard feature of biofuel regulation and sustainability certification.

While attributional LCA is a more tractable exercise than consequential LCA, using only attributional LCA to assess policy may result in ignoring important policy consequences. Perhaps the most important in the case of biofuels is that attributional LCA allows the use of land to be treated as 'carbon free', even though we know that expanding agriculture results in significant land use change GHG emissions. At the level of a batch of biofuel, an attributional result identifying no land use change emissions may be completely correct on its own terms. At the level of a biofuel policy, estimating overall emissions by summing the

attributional results for all the batches supplied and counting no land use change emissions gives a reliably incorrect characterisation of the net policy impact.

If we source biofuel feedstock from farms that have been cultivated for generations, an attributional analysis can identify that there has been no land use change and record zero land use change emissions. Across the whole economy, however, we know that we cannot deliver millions of tonnes of biofuel feedstock without increasing feedstock production somewhere. Indirect land use change emissions, abbreviated as ILUC, are the emissions from land use change that we expect to happen somewhere in the world when we increase demand for biofuels. To estimate ILUC emissions we must turn to consequential LCA tools.

The main tools that have been turned to the question of ILUC analysis are equilibrium economic models. An equilibrium model is a system of mathematical equations representing production and consumption of various goods and services – for ILUC analysis, we focus on doing this for the agricultural sector. The equations are set up so that everything in the economy is in equilibrium – the prices on goods and services are such that everything that is produced is consumed. To use such a model to assess ILUC emissions, we simply move one or more of the values in the model out of that equilibrium state – for ILUC modelling that generally means assuming an increase in biofuel consumption in some region. Having disturbed the model, supply and demand are no longer in balance. The increase in biofuel consumption means an increase in feedstock demand, which implies an increase in feedstock prices, which implies increases in feedstock production and feedstock imports, which can drive increases in production and trade of related goods, which may require expansion of the land dedicated to agriculture. The mathematical model is allowed to settle into a new equilibrium, in which the total area of land farmed will have increased. That increase, caused in the model by the increase in biofuel demand, is ILUC, and if we can put a number on the carbon stock change associated with that land use change then we can calculate ILUC emissions.

Equilibrium ILUC models fall into two categories: partial equilibrium models in which only the agricultural sector is modelled, and general equilibrium models in which the whole economy is modelled. Partial models allow greater detail, but general models allow a wider scope of analysis. Both types of model have been used to assess ILUC emissions, and the results of such ILUC modelling exercises have provided evidence that ILUC emissions are potentially large compared to the GHG benefits that biofuels might deliver by displacing fossil fuel use. The uncertainties in these modelling exercises are considerable. Modelling the global agricultural economy is a fundamentally difficult task, and any of the many assumptions and simplifications made in the modelling could be challenged and debated. Nevertheless, these tools represent the best available evidence that we have about the likely magnitude of ILUC.

In the European Union, two modelling exercises undertaken for the European Commission have defined the discourse on expected ILUC emissions. The first of these was general equilibrium modelling by the International Food Policy Research Institute with the MIRAGE model; the second was partial equilibrium modelling by the International Institute for Applied Systems Analysis with the GLOBIOM model. While there are important differences between the results from the two models, there are also two important similarities. Both models find that ILUC emissions from ethanol feedstocks (starchy and sugary crops) are likely to be significant but unlikely to eliminate the GHG benefit from the use of ethanol. Both models find that ILUC emissions from vegetable oils are likely to be large enough to eliminate most or all of the climate benefit of using biodiesel. Perhaps the most striking difference between the results of the two models is that the GLOBIOM modelling concluded that ILUC emissions from the use of palm oil and soy oil may be very high indeed – enough to not only eliminate the climate benefit of biodiesel use, but to make biodiesel mandates a significant driver of climate change.

Confronted by these conclusions, the European Union has made major changes to its biofuel policy. Firstly, it has placed a cap on the support given to food-based biofuels. Where the first RED (RED I) treated expansion of the food-based biofuel industry as a goal, the recast RED (RED II) sees the food-based biofuel industry as an interim step to be moved beyond. Secondly, new support measures have been introduced to encourage the development of advanced biofuel technologies that can allow the use of materials as feedstock that we do not expect to be associated with indirect land use change. Thirdly, an assessment of 'ILUC-risk' has been introduced, with feedstocks identified as high ILUC-risk being excluded from access to national subsidies. The ILUC risk assessment reflects a compromise between the recognition that action on ILUC is needed, and a caution about relying directly on the numerical results from consequential modelling as a regulatory tool. It involves identifying which crops are directly associated with the conversion of high carbon stock areas, and the assumption that where this direct link is strong ILUC emissions are likely to be highest. The initial ILUC-risk assessment identified palm oil as a high ILUC-risk feedstock, and EU Member States have already begun to adopt measures to remove subsidies from palm-oil-based fuels.

These three measures go a long way to reorient EU biofuel policy, but Member States are also given the leeway to consider taking additional measures to increase support for biofuels believed to cause less ILUC, or reduce support for biofuels believed to cause more ILUC. Article 26(1) of the RED II allows Member States to,

> *Distinguish … between different biofuels, … produced from food and feed crops, taking into account best available evidence on indirect land-use change impact.*

This article is newly introduced in the RED II, which is only now being implemented by Member States, and there has not yet been time to explore how far this legal leeway to distinguish fuels goes. The RED II gives as an example the option to limit the use of vegetable oils for biofuel more than the use of starchy and sugary materials, but if the best available evidence on ILUC supports the conclusion that some feedstocks within these categories have higher ILUC than others, there is no obvious legal barrier to implementing a more tiered system of support.

There are three pieces of evidence on ILUC that can clearly be identified as constituting (in the eyes of the European Institutions) elements of the best available evidence. These are the modelling studies with MIRAGE and GLOBIOM, and the high ILUC-risk assessment. A review for the European Commission led by Wageningen Economic Research of the evidence base on ILUC identified a number of other studies in the literature that could also be considered relevant. In the absence of further guidance from the Commission, Member States must decide for themselves how to balance this body of evidence, and whether they consider it appropriate to regulate on the basis of numerical values drawn from it. Taken as a whole, this evidence set provides a clear basis to consider reducing support for vegetable oils in general, and for palm and soy oil in particular. Denmark already plans to extend the required phase out of support for palm-based fuels to cover soy-based fuels as well.

Beyond the highest ILUC feedstocks, there are various ways that the findings from ILUC research could be applied in regulation to further distinguish between different biofuels. One approach that is already anticipated in the RED II would be to impose a cap setting a lower limit on use of food-oils (primarily rapeseed and sunflower) to meet fuel supplier obligations. Imposing such a limit could create a two-tier market in biotickets under a Danish GHG reduction obligation, reducing the value of biotickets from food biodiesel.

A more complex approach would be to follow the example of the Low Carbon Fuel Standard in California, where levels of support for biofuels are determined by GHG emissions savings calculated by comparing a hybrid LCA score (attributionally calculated direct emissions plus consequentially calculated ILUC emissions) to a fossil fuel comparator. Under such a system, the support provided to vegetable oil-based biodiesel and first-generation ethanol would be reduced based on the estimated ILUC emissions, which could be determined based on consideration of one or more modelling exercises. Under this system, food-oil based fuels would not be subjected to a hard limit on the support available but would be put at a clear disadvantage in the market. While this approach has been somewhat successful in California, it has been rejected at the EU level and Recital 81 of the RED II states that ILUC, "cannot be unequivocally determined with the level of precision required to be included in the greenhouse gas emission calculation methodology". While there is a strong legal argument to be made that this recital does not restrict the regulatory leeway given to Member States under Article 26(1), we would expect the European Commission to firmly discourage Member States from adopting such an approach.

A complementary measure would be to create a system of additional support for biofuels certified as low ILUC-risk. The idea of low ILUC-risk certification is to identify biofuel production systems that bring additional feedstock to the market without interfering with existing uses of feedstock materials. The main categories of low ILUC risk project are bringing unused land into agricultural production and delivering productivity increases on land that is already farmed.

Currently the main role for this certification in the RED II is allowing some palm oil producers access to the market despite the high ILUC-risk rules, but this approach could be extended to encourage low ILUC-risk cultivation of other crops. While low ILUC-risk approaches are applicable to rapeseed and sunflower oil, the largest opportunities for low ILUC-risk project development have been identified in regions with relatively large unused land resources or with low yields due to a failure to optimise production, and therefore opportunities for project development in Denmark itself may be limited.

There is no question that ILUC is a challenging area to analyse, and a challenging area to regulate. The considerable uncertainties in ILUC analysis and disagreements about ILUC in the stakeholder community mean that active engagement is of paramount importance, especially if fuel suppliers are to be asked to deliver compliance under new regulatory requirements. Nevertheless, the flexibility granted by Article 26(1) creates an opportunity to use additional regulatory tools to further limit ILUC emissions and maximise the climate benefit from its biofuel support policy. An approach combining additional limits on food-oils based fuels with support for low ILUC-risk projects would significantly improve the climate performance of Danish transportation and should be acceptable to the European Commission.

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Glossary

AEZ	Agro-Ecological Zone
CAEP	ICAO Committee on Aviation Environmental Protection
CARB	California Air Resources Board
CGE	Computable General Equilibrium
CORSIA	Carbon Offsetting and Reduction Scheme for International Aviation
DDGS	Dried distillers' grains and solubles
DGS	Distillers' grains and solubles
EEA	European Environment Agency
EISA	Energy Independence and Security Act
EPA	U.S. Environmental Protection Agency
FAO	Food and Agriculture Organisation of the United Nations
FQD	Fuel Quality Directive
GHG	Greenhouse gas
ICAO	International Civil Aviation Organisation
IFPRI	International Food Policy Research Institute
IIASA	International Institute for Applied Systems Analysis
ILUC	Indirect land use change
IPCC	Intergovernmental Panel on Climate Change
JEC	JRC-EUCAR-CONCAWE
JRC	Joint Research Centre of the European Commission
LCA	Lifecycle analysis
LCFS	Low Carbon Fuel Standard
LUC	Land use change
LULUCF	Land use, land use change and forestry
OECD	Organisation for Economic Cooperation and Development
PEF	Product Environmental Footprint
RED I	The first Renewable Energy Directive, covering the period 2010-2020
RED II	The recast Renewable Energy Directive, covering the period 2021-2030
RFS	The U.S. Renewable Fuel Standard
UCO	Used cooking oil

1 Introduction

For many years, EU Member State governments and the European Commission have supported the supply of biofuels through a variety of mandates, targets, grants and favourable tax treatment. Looking back to the 90s and 2000s, biofuel policy was framed as meeting three broadly co-equal objectives – supporting rural incomes, promoting energy security and contributing to climate change mitigation. As time has passed and the sense of urgency in climate policy has grown, climate change mitigation has increasingly become foremost among these objectives, so that in Europe biofuel mandates are now understood primarily as climate change policy.

As climate change has become increasingly central as a motivation for the introduction of biofuel targets, concerns about the climate change impact of biofuel use have taken a central place in the policy debate. There is a basic understanding that biofuels can be presumed to have low CO_2 emissions because they are renewable. Carbon accounting rules developed for the implementation of the Kyoto Protocol treat biomass combustion as if it had zero CO_2 emissions at the point of combustion – following that carbon accounting convention allows the exhaust pipe CO_2 emissions from vehicles running on biofuel to be ignored. There are two basic premises that inform this carbon accounting simplification. Firstly, it is observed that carbon in plant matter is formed by absorption of CO_2 from the atmosphere. There is therefore a sense of a cyclic element to biomass energy – if we absorb CO_2, then release it again, and then in due course absorb it all over again, there is no net change in atmospheric CO_2 concentrations. This is only true, however, if the process of biomass production is truly in such a cyclic state – if we harvest biomass for energy and then it is not grown back, then net changes in atmospheric CO_2 can still occur.

The second plank of the zero-accounting convention is that emissions from net carbon stock changes still get accounted, just elsewhere in the GHG inventory. In the Kyoto rules, changes in land carbon stocks are recorded in the land use land use change and forestry (LULUCF) inventory instead of the industrial inventory. This division of inventories has some appeal in theory, but it can be problematic in practice. If national CO_2 targets do not include LULUCF emissions, then biomass energy use could see a form of leakage whereby CO_2 emissions are moved out of more regulated sectors only to show up in LULUCF where they are not limited. Similarly, if policies such as cap and trade or renewable energy mandates that apply only to certain sectors create higher carbon prices[1] for industrial and transportation emissions than LULUCF emissions, we can create an economic incentive to simply move CO_2 between inventories rather than reduce emissions in absolute terms.

In the 2000s, as interest in biofuel policy grew in both North America and Europe, so did concern that it was inappropriate to act as if biofuels were fundamentally carbon neutral. It was recognised that the production of biofuels requires energy inputs that in many cases are of fossil origin – natural gas for heat and power, and diesel fuel for agricultural equipment. It also requires the use of agricultural chemicals, including nitrogen fertilisers that can lead to nitrous oxide emissions with a high global warming potential. Regulatory treatments therefore developed that included lifecycle analysis (LCA) requirements so that GHG emissions associated with biofuel production systems could be taken into account. The original Renewable Energy Directive ("RED", European Union, 2009a) introduced a

[1] Renewable energy policy generally does not directly impose a carbon price, but the value created by incentives or penalties in such policies can be thought of as creating an implied carbon price.

requirement that biofuels must have a lifecycle GHG emissions intensity at least 35% below the lifecycle GHG intensity calculated for petroleum-based fuels, while a requirement (Article 7a) was added to the Fuel Quality Directive (FQD, European Union, 2009b) requiring that the average GHG intensity of the transport fuel supplied in Europe should be reduced by 6% on a lifecycle basis in 2020 compared to a 2010 baseline.

The introduction of LCA requirements seemed to be a basis to resolve concerns about the production of biofuels, but did not confront the question of whether there was a carbon opportunity cost associated with turning large areas of agricultural land over to biofuel production. To put it another way, policy makers had failed to adequately address the question of how the agricultural system would deliver biofuel feedstock without taking it away from food consumers. It was not that the question of land use had not been considered at all. For example, the European Environment Agency (Wiesenthal et al., 2006) assessed the potential to produce biomass for energy in Europe in an environmentally sustainable way. Such analysis, however, modelled the agricultural system as it could be in an idealised scenario – not as it could reasonably be expected to respond to economic incentives. Wiesenthal et al. (2006) envisioned a domestic bioenergy market that would not rely on feedstock imports and that would transition from first generation biofuels to 'cellulosic' biofuels by 2020. Cellulose and ligno-cellulosic biomass are the families of chemical compounds that constitute most of the non-edible parts of plants, including grassy and woody material, leaves, straw and stalks. These cellulosic materials have lower value than food and feed commodities[2], and because cellulosic and ligno-cellulosic material is available as residues from agricultural and forestry activities and can be produced on lower quality land, using these materials for biofuel feedstock can be expected to have lower land use impacts.

The sense that the carbon opportunity cost of biofuels had not been properly taken into consideration was crystallised with the release of two reports at the end of the 2000s. Fargione et al. (2008) showed that the carbon debt from many specific land use changes would eliminate the climate benefits from producing biofuels on newly farmed land. The discussion was permanently transformed, however, by Searchinger et al. (2008), which presented economic modelling results suggesting that the land use changes necessary to accommodate growing biofuel demand could eliminate the expected GHG benefits from a U.S. corn ethanol mandate. The field of indirect land use change (ILUC) modelling and the estimation of ILUC factors (estimates of the GHG emission from land use change associated with producing a unit of biofuel) was born. The results presented in Searchinger et al. (2008) have been highly controversial and subject to much criticism from supporters of the biofuel industry, but nevertheless shifted the conversation in both North America and Europe so that it was no longer viable for policy development to take land availability for granted.

ILUC has sometimes been represented by critics as 'just a theory', but the question posed by Searchinger et al. (2008) and answered in subsequent ILUC modelling emerge from a simple consideration of conservation of mass. The biofuel industry uses large quantities of agricultural commodities, and this material has to come from somewhere, either by increasing total agricultural production or by shifting consumption. On the demand side, biofuel feedstock could be made available by reducing the amount of material consumed as food for people, feed for animals, or feedstock for industrial applications (for example vegetable oils in beauty products). On the supply side, biofuel feedstock could

[2] Though noting that ruminant animals including cows can digest cellulosic fibre in grass and hay, often referred to as forage in the livestock feed context.

be made available by farming new areas or by producing more material on areas that are already farmed.

ILUC modelling is about building scenarios for the balance between these supply and demand side responses, and assigning estimated GHG emissions consequences to them. There is a wealth of evidence at this point to confirm that these GHG emissions are potentially significant and must be considered when developing biofuel policy. In the period since (Searchinger et al., 2008) was published, there have been numerous attempts to model ILUC. One review for the European Commission (Woltjer et al., 2017) found over 100 studies providing quantitative information. In this report, we discuss some of the tools have been developed to model these ILUC-related GHG emissions, and then discuss the policy approaches that could be available to the Government of Denmark to maximise the GHG benefits and minimise the externalities of meeting its renewable energy targets.

The acknowledgement of indirect emissions is a challenge for attempts to create 'performance based' regulations. Performance based climate policy aims to reward (or penalise) systems in proportion to their climate impact. Basing support on measured performance is seen as a way to deliver technology neutral policy, allowing quite different technology options to compete in a compliance market based on their ability to deliver GHG reduction. In the transportation fuel space, proposed changes to the RED (European Commission, 2021b) would move the EU's main target for renewable energy in transport from an energy basis to a GHG performance basis, and similarly Denmark expects to shift its main national renewable fuel targets from an energy basis to a GHG basis by 2024. As we discuss in more detail below, ILUC emissions are not currently reflected in the LCA requirements set in the RED, and this means that biofuels for which we expect large ILUC emissions may still score well on the performance metric of the Directive.

Implementing a performance based regulatory framework based on a partial characterisation of performance risks undermining the technology neutrality of the system. In 2008 the UK Government's Gallagher review of the indirect effects of biofuel production (RFA, 2008) warned that under the current carbon accounting framework, "GHG-based targets may result in a greater land requirement, and land-use change, than a volume or energy-based target". Excluding indirect emissions from the performance assessment will lead to biofuels with high ILUC being unfairly advantaged compared both to biofuels with lower ILUC and to renewable fuels of non-biological origin[3] for which no ILUC emissions are expected.

In this report we discuss policy options to manage ILUC emissions from biofuels, some of which would help to reduce these distortions to technology neutrality in a system based on GHG reduction targets.

[3] Renewable fuels of non-biological origin (RFONBOs) are fuels produced primarily by chemical synthesis from hydrogen that is produced from electrolysis powered by renewable electricity.

2 Frameworks for considering land use change

The assessment of ILUC has been described as a "wicked problem" (Palmer, 2012), where the complex interplay of interdependent factors makes it difficult or impossible to come to a generally agreed solution. Land use changes are the result of evolving interactions between government policy, market conditions and individual decision-making; hence it is far from trivial to quantify the expected impact of a given biofuel policy. At a high level, approaches to quantifying the risk of ILUC can be divided into two categories: "consequential" and "attributional". In consequential analysis we try to draw conclusions about what consequences will follow from a given decision (e.g. a decision to increase a biofuel mandate). In attributional analysis we try to attribute known environmental impacts across a defined set of activities.

2.1 Lifecycle analysis and lifecycle analysis questions

Lifecycle analysis of GHG emissions is the discipline of identifying the GHG sinks and sources within a defined scope that are associated with a given activity. The definition of what counts as associated depends on the question that the lifecycle analysis is intended to answer. Consider a simple example relating to vehicle emissions. If we ask the question, "How much carbon dioxide is emitted from the tailpipe of this vehicle?" this sets the narrowest possible scope of analysis – measuring a single source. If instead we ask a question like, "How much carbon dioxide is emitted by this vehicle and by all the processes required to produce the fuel for this vehicle?", this immediately sets a much wider scope of analysis. Lifecycle analysis results can only be properly understood with a precise understanding of the question that is being asked, and therefore of the scope of the analysis.

In the example above, neither of the questions being asked is right or wrong – they are both sensible questions to ask in the right context. In the context of a vehicle efficiency standard, it might be entirely appropriate to focus only on the emissions from the tailpipe. In the context of biofuel regulation, focusing only on the tailpipe emissions would be pointless because the combustion of ethanol or biodiesel releases about the same amount of carbon dioxide as the combustion of petrol or diesel does.

What then is the lifecycle analysis question that we do want to ask when considering biofuel policy? At the policy level, we might want to ask, "What is the expected change in net global emissions if we require the supply to the transport of an additional unit of biofuel?" Answering such a question could inform a decision about the benefits that can be achieved by mandating biofuels use. If instead we were focused on understanding the efficiency of a specific biofuel production process, we might ask, "What emissions are associated with the processes required to produce a unit of biofuel by growing a given feedstock?" The first question is a consequential question – it asks what the consequence of a given change would be. The second question is an attributional question – it asks how we can attribute emissions to a given system.

2.2 Consequential and attributional

Consequential approaches can be thought of as having a forward-looking character. While the approach of consequential analysis is forward looking, consequential models are not only applied to future events. Consequential models can also be utilised with a view to better understanding outcomes that have already happened, but where it is difficult or impossible to directly observe the causal relationships involved. Analysis of biofuel mandates that have already been introduced is an example of this. In a consequential approach to assessing indirect land use change, an analyst asks a question such as, "If government policy is used to expand biofuel demand, what amount of additional land would we reasonably expect to be brought into agricultural production in order to allow that demand to be met?"

Answering such a question requires the analyst to develop models of the way that the supply of agricultural commodities changes as a consequence of changes to demand, and of the factors that determine where new land is brought into production. A given set of assumptions about how the system might respond forms a scenario, and consequential modelling exercises often report the results for several scenarios based on varying assumptions. The scenario results can be thought of as representing plausible outcomes, or if we feel confident that we have a good model for system responses we might consider a 'central' scenario result as a prediction. Consequential modellers are sometimes uncomfortable with the language of prediction, as they are generally cautious of claiming to be able to accurately predict the future and recognise that there are uncertainties throughout any consequential assessment. Another way to think about the scenarios from a consequential is that even if we don't expect a specific scenario to 'come true', analysing scenarios provide a reasonable basis by which to set our expectations about the scale of impacts that are likely to happen.

Attributional approaches can be thought of as having a backward-looking character. In an attributional approach, an analyst asks a question such as, "Given that some land use changes have been observed in a defined period, what fraction of these land use changes should be attributed to changes in biofuel demand rather than to other drivers?" Answering this type of question requires the analyst to identify where land use changes have occurred and to develop models for attributing land use changes between different drivers of land use change.

Consequential approaches can also be thought of as being more oriented towards assessing "marginal" impacts of making changes while attributional approaches are more oriented towards assessing "averaged" impacts of some set of processes within a system in equilibrium. The marginal emissions impact of a given change can be defined as the sum of the emissions increases and reductions occurring in the wider system as a result of making that change, divided by the size of the change. The averaged emissions impact of some set of processes can be defined as the sum of all emissions sources and sinks associated with those processes divided by the quantity of outputs produced by those processes.

In a world of high-level climate targets and national greenhouse gas emissions inventories, one of the appealing features of attributional approaches is that they can be structured in such a way that all emissions in an inventory are allocated to one and only one economic activity. If we set consistent rules defining the system boundary for emissions calculations, then all of Denmark's GHG emissions could in principle be divided neatly among Denmark's economic operators and citizens without double counting any tonne of carbon dioxide. Consequential approaches do not support this type of allocation of emissions across agents, and therefore can feel inconsistent with the inventory approaches at the heart of global climate policy.

2.2.1 Average versus marginal emissions – refinery example

To help explain the difference between marginal and average impacts, consider a fictionalised[4] example in the fossil fuel supply chain. In a simple refinery, oil is put through a distillation column to separate it into fractions according to the boiling point. The hydrocarbons with the lowest boiling point are separated off and will be upgraded for sale as transport-grade petrol. This is followed by "mid-distillates" that are suited to be upgraded into transport grade diesel fuel, and then the remainder can be used as fuel oil. The distillation column has relatively low energy use, and therefore the CO_2 emissions from this simple refinery are low – let's say that it produces 5000 tonnes of fuel and emits 500 tonnes of carbon dioxide per year. The average emission per tonne of fuel output is 0.1 tonne of CO_2 per tonne of product.

Imagine now that we add a new refinery unit, a hydrocracker, which can be used to convert some of the fuel oil into transport-grade diesel fuel. Hydrocrackers require hydrogen as an input and are much more energy intensive than the distillation column. Let's assume that the hydrocracker allows the refinery to produce an extra 400 tonnes of diesel and 400 tonnes less fuel oil, but also increases the refinery's CO_2 emissions by 250 tonnes. If we take an average emission for the whole refinery, we now have 750 tonnes of CO_2 to produce 5000 tonnes of output, an increased average emission of 0.15 tonnes of CO_2 per tonne of product. We could also look at the marginal emission cost of producing the additional diesel fuel. Producing 400 tonnes of extra diesel caused emissions to increase by 250 tonnes, so the marginal emissions from producing additional diesel were 0.625 tonnes CO_2 per tonne of product. The marginal carbon intensity of extra diesel production is four times higher than the average carbon intensity.

Neither of these results is more technically 'correct' than the other, and both results could be misleading if taken out of context. The average result tells us more about the overall emissions from the refinery than the marginal result does – if we assumed that every tonne of diesel produced cause 0.625 tonnes of CO_2 production, we would overestimate the total emissions because the marginal value is only applicable to the extra fuel put through the hydrocracker. If we want to attribute the overall emissions across the product pool, we will tend to use some variation on the average approach. In real lifecycle analysis we might want to apply some different weighting when we allocate the overall emissions to the different fuels, for example allocating a larger fraction of the emissions to the most valuable products rather than dividing them up by mass. This is just a slightly more sophisticated form of attributional analysis.

Just as the marginal result doesn't tell us anything useful about total emissions at the refinery, so the average value tells us very little about the increase in emissions we should be expecting if we decide to produce additional diesel. The marginal number is a result of consequential thinking, because it tells us about the consequences of changing something about the refinery configuration. Note that the JEC well-to-wheels study for the European Commission (Prussi et al., 2020) provides marginal

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[4] I.e. the numbers used are illustrative only, and are not intended to be realistic values.

estimates of the emissions associated with refining an additional quantity of each petroleum-based fuel in the EU refinery complex. This decision was taken because the well-to-wheels results are intended to tell us something about the emissions cost or benefit of changing the fuels we produce – the lifecycle analysis question requires a consequential answer. It is also worth noting that the marginal result is much more sensitive to the initial configuration of the system than the average result is. If the refinery is already at its maximum 'basic' diesel capacity, then producing more diesel will involve using the hydrocracker. If, however, you had a starting configuration where some fuel molecules suitable for diesel were being sold as fuel oil because of a lack of diesel demand, then diesel production could have been increased without using the energy-intensive process, and the marginal result would have been completely different.

2.2.2 Consequential and attributional views of land use change

In the context of land use change, attributional and consequential approaches tend to give quite different answers. The most common attributional approach to land use change accounting for biofuels is to say that land use change emissions should be attributed to batches of biofuels produced on areas of land where a land use change is known to have occurred within some defined period of time. Under the RED, it is required to include land use change emissions in the attributional calculation if the feedstock comes from land that has changed use at some point since January 2008[5]. As land use change emissions for conversion of grassland or forestland to cropland are generally so great as to make it impossible for a batch of biofuel to meet minimum GHG saving criteria, the upshot of this rule is that the biofuels produced in Europe tend to be from feedstock batches that can be associated with areas of land under long-term crop production. Feedstock batches from land that has recently changed use get supplied to other less-regulated markets.

Attributional approaches can also be developed to attribute emissions not only to areas of land where land use change has actually occurred. This could involve defining some system for attributing historical land use change emissions across units of feedstock production including on pre-existing agricultural land. A simple version of such an approach would be to assess all land use change emissions associated with agricultural expansion in a given region and period, and to attribute those evenly (weighted by mass, value or some other relevant characteristic) to the agricultural production in that country in the same period. As discussed in section 4.2, the high ILUC-risk assessment undertaken for the RED II can be thought of as a more sophisticated version of such an attributional approach, in which emissions are attributed by units of additional production rather than to all production.

Consequential approaches, in contrast, attempt to characterise the amount of land use change that we would expect to occur as a result of some defined increase in biofuel use. Consequential models are often informed by historical information, and so the sort of attributional exercise with historical data described above could be used as an input to the development of a consequential model. Where attributional approaches generally draw simple links (e.g. "this land use change emission and this feedstock production happened in the same geographical location, so we will treat

[5] Strictly, the date for assessing land use changes is "January 2008 or 20 years before the raw material was obtained" – but in practice January 2008 remains the cut-off until the year 2028.

them as being connected) consequential approaches require more complex links (e.g. "consumption of additional vegetable oil for biodiesel would raise vegetable oil prices, which would create an incentive for investment in expansion of rapeseed oil area in Europe and palm oil area in Indonesia, which would result in net land use change emissions"). Very simple attributional approaches tend to be more analytically precise – if an area of forest is converted to wheat production and the wheat is supplied to an ethanol refinery, this can be objectively established and there is no denying that the associated emissions must be reported under the RED II. Consequential approaches (and more complex attributional approaches) require more decisions about how systems work and how to attribute responsibility. These decisions are always partly subjective, and therefore are constantly controversial.

2.2.3 Consequential and attributional approaches for the whole lifecycle

We can also consider consequential versus attributional approaches for the production emissions for biofuels. The regulatory lifecycle analysis used to set typical emissions values for biofuels in the RED has an average-attributional character. In the emission factors provided in the Directive, if a tonne of corn is processed for biofuel feedstock it is assumed that consumption of various inputs (fertiliser, farm energy, pesticides) is consistent with producing that much corn on an additional area of typical land following typical agricultural practices. Alternatively, it is permitted to identify the actual consumption of inputs on a specific farm from which feedstock is sent to the biofuel producing facility.

The RED does not consider, however, that the production of the additional corn needed as biofuel feedstock might not be best described by considering a discrete area of corn production at a single farm. The additional corn might, for example, have been produced by increasing fertiliser application across a wider area of already farmed land. In this case, the marginal impact could be assessed based on the increase of fertiliser use, with some additional tractor fuel consumption but with no additional use of pesticides or seeds. Equally, the RED does not consider the possibility that extra corn could be made available for biofuel production by replacing corn in the animal feed market with additional barley, in which case a marginal analysis of corn ethanol might need to consider the emissions of that additional barley production. The RED differs in this regard from the Renewable Fuel Standard in the United States, where agricultural emissions for biofuel production have been estimated using consequential tools.

Figure 1 and Figure 2 provide an illustration of the difference between attributional and consequential LCA approaches. In Figure 1 the attributional approach is represented. In the illustration, the circles at the top represent the sum of all emissions in the global system. The dark sectors in the circles below represent the land use change, farm inputs and processing emissions directly associated with the processes that produce a batch of biofuel, and on the right we see that standard attributional analysis would assume that this produced biofuel displaces fossil fuel production on a 1:1 basis.[6] If there are

[6] Not that this assumption of 1:1 fossil fuel displacement is not intrinsic to the attributional approach, which could be used to assess the biofuel production process without making reference to fossil fuel displaced. The assumption is so normalised as a part of attributional biofuel LCA, however, that we have included it here for the illustration.

no land use **changes** associated with the area where the biofuel is grown, then no land use emissions would be attributed to it.

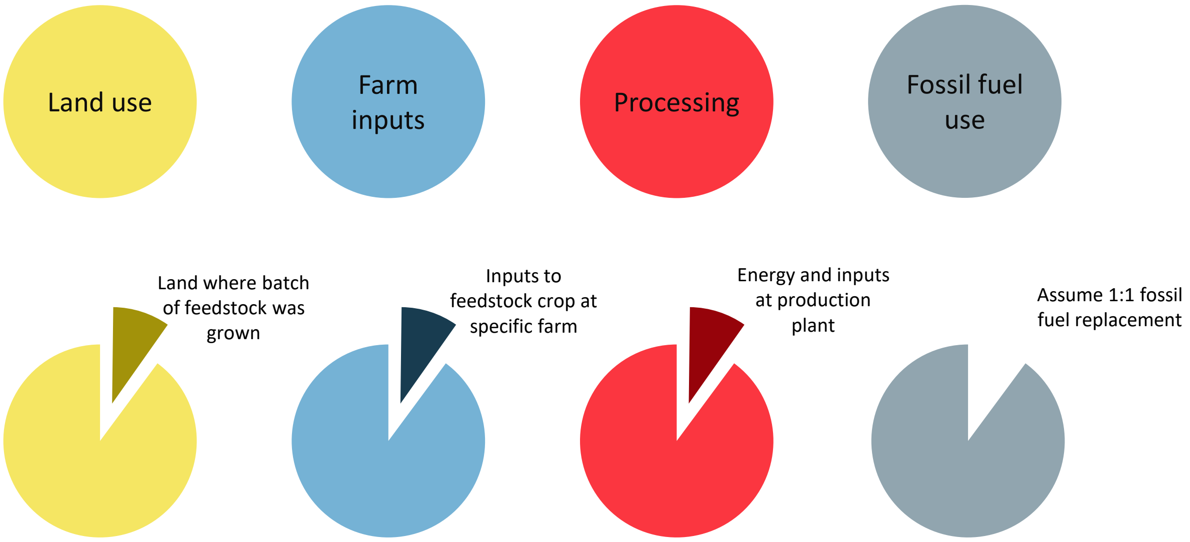

Figure 1 Schematic illustration of attributional lifecycle analysis

Figure 2 shows the consequential case. Now, the circles at the top represent the baseline global system **before** the introduction of additional biofuel demand. Below that is a schematic illustration showing that in consequential analysis additional emissions may be identified remotely from the sites identified as directly supplying biofuels. The location of emissions that would be identified as associated with the biofuel production process under an attributional system is indicated by the hatched areas. Whereas the attributional approach associates a 'slice' of emissions from the system with the biofuel production system, the consequential approach looks for areas in which there are new emissions compared to the baseline. It is possible that there might be no overlap between the emissions sources identified under a consequential analysis and those identified under an attributional analysis for the same notional batch of fuel.

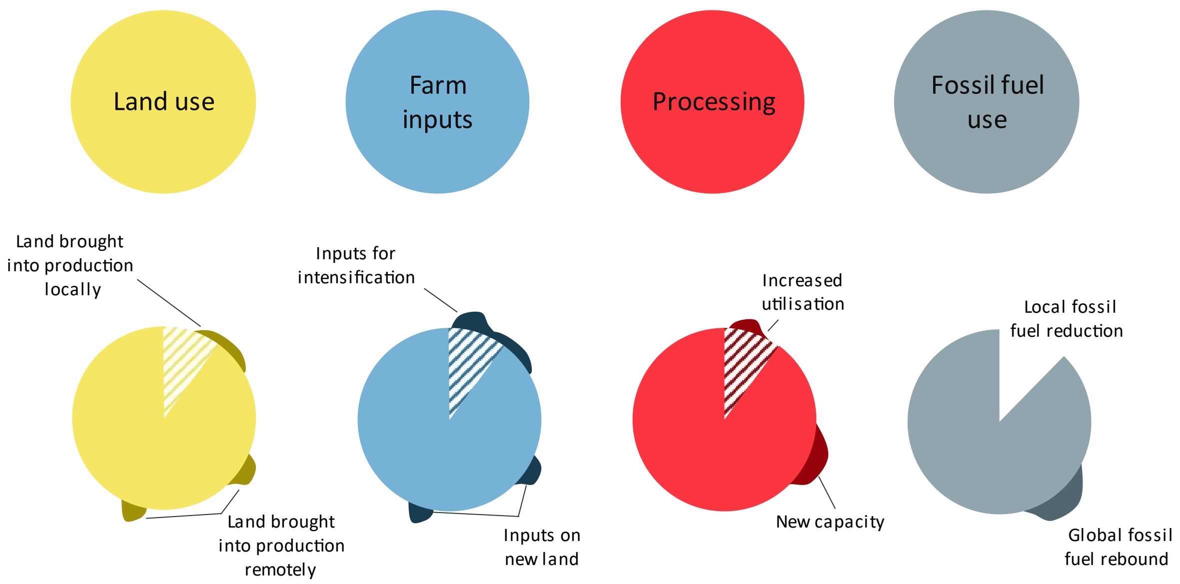

Figure 2 Schematic illustration of consequential lifecycle analysis

Figure 2 depicts the case in which a consequential approach is adopted for all elements of the lifecycle. However, in practice it is common to adopt a hybrid approach, for example using consequential reasoning only for ILUC and attributional analysis for the farm and factory. This is discussed further in section 2.5.

2.3 Comparing lifecycle analysis approaches

There is no single "right" way to deal with questions in lifecycle analysis. This is true for assessing ILUC-risk just as it is true for assessing oil refineries or solar panels. Some of the decisions made in setting up an LCA system will always require value judgments and trade-offs. Good practice requires that these decisions are acknowledged and explained, maintaining focus on the objectives of the exercise.

Having decided what lifecycle analysis question one would like to answer, one must decide whether consequential or attributional tools are the most appropriate. There is no clean dividing line between these tools. Consequential models may include elements that are informed by attributional analysis, and attributional models may include elements that are informed by consequential thinking. For example, in some consequential ILUC models, attributional analysis of historical land use changes in Southeast Asia has been used to develop assumptions about the likely impact of additional palm oil demand on peat clearance. In the attributional analysis that underpins the high ILUC-risk assessment made under the recast RED, consequential ideas are used in allocating observed deforestation between agricultural commodities, livestock farming and timber production.

While both consequential and attributional approaches have their place, the question most relevant to assessing the likely ILUC impacts of biofuel policy is a consequential one which can be stated as, "If we use policy measures to increase consumption of biofuels within our jurisdiction, then what is the expected consequence in terms of the net change in emissions from land use changes?" This is quite distinct from the sorts of attributional questions that could be analysed, such as, "What land use change emissions have happened in the last five years on the land where the feedstock processed in this biofuel plant was grown?"

2.4 Approximately right or precisely wrong?

When choosing analytical systems to assess environmental impacts, there is a risk of falling into an approach that has been referred to as the 'streetlight effect' or 'drunkard's search'. These terms are coined in reference to an old joke-cum-parable, which goes something like this (David Freedman, 2010):

> *A policeman sees a drunk man searching for something under a streetlight and asks what the drunk has lost. He says he lost his keys and they both look under the streetlight together. After a few minutes the policeman asks if he is sure he lost them here, and the drunk replies, no, and that he lost them in the park. The policeman asks why he is searching here, and the drunk replies, "This is where the light is."*

In the context of choosing lifecycle analysis approaches for biofuel policy, the equivalent of looking under the light for keys that are not there is to use attributional models that offer relatively precise results when needing to answer consequential questions. Uncertainties in attributional LCA relate primarily to technical questions that could be resolved with improved data, and therefore attributional LCA fits our expectations of a scientific exercise. If there is uncertainty about typical rates of fertiliser application by farmers, we can try to improve the answers by undertaking farm surveys and model the response of crop to fertilisation. If there is variability between farms, then in principle we could analyse each farm individually to improve the accuracy of our analysis. Attributional LCA offers answers that feel precise and objective, and therefore to policy makers attributional LCA can appear to be a solid basis for regulatory action[7].

Consequential analysis, in contrast, is afflicted by uncertainties that are much harder to resolve through technical analysis. Consequential analysis requires assumptions about how people behave, about what changes to agricultural systems are possible, about whether farmers are more willing to adopt new practices or farm new land. The questions in consequential analysis are also amenable to further investigation, but the answers don't feel as precise – we may say with confidence that the average corn farmer in Sweden applies 40 kg of nitrogen per hectare per year, but be more cautious to say that the average farmer in Sweden can be expected to increase fertiliser application by an additional 2 kg nitrogen per hectare per year in response to a 10% increase in the price of grain. Attributional lifecycle analysis is at its heart a question of inventory keeping, which feels precise, whereas consequential analysis is a question of predicting behaviour, which is the subject of science fiction stories[8]. Consequential analysis therefore feels more subjective than attributional analysis does, and policy makers tend to be much more reluctant to base regulatory action on specific results from consequential models.

This sense of reluctance to use consequential modelling results directly in regulation is not simply a prejudice of regulators. Even the analysts responsible for modelling ILUC emissions using consequential tools (the models described in more detail in the next chapter) have sometimes also been circumspect about having their results given direct regulatory application through ILUC factors. For example, Laborde (2011) comments that,

> *"Defining crop-specific iLUC appears to be quite challenging, both from a modelling point of view (uncertainties are still large) and from an incentive point of view: how could the soybean producers in South America be considered responsible for the governance of peat lands in Southeast Asia?",*

While the uncertainties in consequential modelling are a problem, it is a non-sequitur to go from identifying uncertainty to deciding not to take any action. Some defenders of crop-based biofuel production have focused on the uncertainty in consequential ILUC modelling, arguing that it is so great that it should not be used to support decision making (see e.g. Sigurd Næss-Schmidt et al., 2019). One lifecycle analysis expert formerly at the EU's Joint Research Centre was known to answer this challenge by saying that it is "Better to be approximately right than precisely wrong"[9]. There is a considerable weight of evidence that indirect land use change emissions are significant, and one does not need to pinpoint them to 3 decimal places to know that they must inform policy making. While

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[7] Although it should be noted that attributional LCA may sometimes offer a false sense of precision – (Plevin, Delucchi, et al., 2014) discusses outstanding uncertainties in attributional results.

[8] Isaac Asimov's Foundation novels describe a far future in which a mathematician has achieved the feat of producing accurate predictions about human decisions.

[9] While often attributed to J M Keynes, this aphorism was coined by Carveth Read in 1898 in the form "It is better to be vaguely right than exactly wrong".

expressing caution about crop-specific ILUC values, Laborde (2011) does not doubt that policy should be informed by the results of modelling,

> *"The strategy should be to limit the overall scope of the mandate or to increase the threshold of eligibility of direct savings for all feedstocks… our evidence shows that different treatments should be used for ethanol (lower risk of large land use emissions) and biodiesel."*

These are not easy questions, nor do they point to easy decisions. It is entirely legitimate for stakeholders to query the use of policy analysis tools that show great uncertainty, and it is legitimate for stakeholders to challenge policy makers to come to firmer conclusions. In the context of biofuel policy, however, the appeal to uncertainty as a basis to ignore ILUC modelling results contains a problematic central fallacy. As was noted above, there is no question that producing crops for biofuels is associated with land use, and there is no question that expanding agricultural land use generally results in land use change CO_2 emissions. This means that where there is uncertainty about ILUC emissions, **there is uncertainty about whether biofuel policy is able to deliver any net climate benefit**. It is difficult to argue that if there is uncertainty about whether imposing costs on the public delivers any benefit the default position should be to continue imposing those costs indefinitely.

Searchinger (2010) suggests that by adopting an attributional framework in which land is treated as having no carbon opportunity cost, biofuel policy has entered a paradigm in which the normal burden of proof has been reversed. We find ourselves in the position that the climate case for supporting biofuels is often taken as a given because of the carbon accounting convention of treating biomass combustion emissions as zero in industrial emissions inventories. Because of this zero-emissions starting point, analysts and stakeholders concerned about the net benefits of biofuel production are challenged to provide evidence to support their concerns. Searchinger (2010) argues that in fact the biofuel industry should be challenged to provide evidence that it delivers additional net carbon removals from the atmosphere compared to a baseline without biofuel production[10]. Credible consequential emissions modelling would be one way to provide evidence that this is true. If, however, it is argued that it is impossible to draw conclusions about ILUC emissions, the implication is that it is impossible to decide whether biofuel policy delivers on its main objective of mitigating climate change. If that was true, then the appropriate policy response would be to refocus on areas of transport policy such as vehicle electrification where the benefits are less contested.

As we discuss in more detail in section 6.2, policy makers in the United States have opted to use consequential results directly in regulations. In the European Union the preference has been to keep an attributional lifecycle analysis and respond to ILUC in other ways.

2.5 Hybrid approaches

While attributional and consequential lifecycle analyses are built on different underlying principles, it is not unusual for individual lifecycle analyses to combine elements of both approaches. These 'hybrid' approaches are generally conceived in order to take

[10] Exhaust emissions of CO_2 are left more or less unchanged when biofuels are combusted, so any net climate benefit has to be delivered by increasing CO_2 removals from the atmosphere or reducing emissions elsewhere.

advantage of the relative precision of an attributional exercise while integrating elements of consequential thinking. Pairing frameworks in this way leads to a degree of formal inconsistency – summing an attributional and consequential result means trying to answer two lifecycle analysis questions at once, and therefore compromising on both. The flip side of this is that a hybrid approach may generate analytical results that can inform decision making in a way that would not be possible from using an approach that was solely attributional or solely consequential.

One example of a hybrid LCA approach arises in considering the emissions implications of the production of co-products or by-products in biofuel production systems. For example, fermentation of corn to produce ethanol results in two main outputs – the ethanol itself, and distillers' grains consisting of the unfermented parts of the grain, such as protein and fibre. A standard attributional approach to handle cases where more than one product is output by a system would be to allocate the emissions from the system partly to one product and partly to the other. If the emissions were allocated equally to each of the two products, then each would be attributed an emission factor equal to half of the total emissions from the process. In practice, we generally do not want to allocate outcomes exactly equally between two products and therefore some sort of weighting will be chosen. As was mentioned above, common ways of attributing emissions to co-products are by mass, by energy content or by financial value. The choice of weighting can make a large difference to the result (Thomas et al., 2015). In a system producing ethanol and distillers' grains, the distillers' grains would be allocated a larger share of emissions on the basis of mass than on the basis of value.

One might, however, feel that such an allocation system is a little arbitrary, especially if the analytical focus is on the ethanol. An alternative more consequential approach to considering the co-product would be to ask how whether the availability of distillers' grains allows emissions to be avoided elsewhere in the system. We might look at the wider agricultural system and conclude that the availability of the distillers' grains for use in livestock feed reduces the need for the production of feed corn and soy meal.[11] Instead of allocating the process emissions from corn and ethanol production between the ethanol and distillers' grains, we would attribute all of those emissions to ethanol as the 'main' product. And then calculate a credit term based on the amount of corn and soy production that we believe the distillers' grains can substitute, using results from an attributional assessment of the GHG intensity of growing each of those crops. This is sometimes referred to as a 'substitution' or 'displacement' approach to co-product accounting.

The justification for using this consequential approach would be to argue that it provides a more meaningful characterisation of the emissions implication of co-product generation, and therefore that adding this consequential element gives a more meaningful characterisation of the 'real' emissions intensity of corn ethanol production. Adopting a substitution approach would allow us to make a useful comparison between two systems using their co-products differently. For example, if distillers' grain allowed corn to be displaced from cattle diets or soy to be displaced from pig diets[12], and soy is assessed as having higher production emissions than corn, we might conclude that it is preferable in

[11] In an economic model this consequential logic is extended by considering not only that the availability of co-products allows other feed ingredients to be substituted, but that this interaction will result in adjusted prices for these feed commodities which could in turn affect other decisions – for example allowing expansion of the livestock sector by reducing feed costs.

[12] Note that this is a simplification to illustrate the point.

emissions terms to build ethanol plants in regions where only pigs are raised than regions where only cattle are raised.

In the context of land use change, a hybrid approach can be adopted by adding the result of a consequential assessment of ILUC emissions to an attributional assessment of feedstock and fuel production emissions. Such an approach is taken under the California LCFS, combining ILUC factors calculated consequentially using GREET with process emissions calculated attributionally using the CA-GREET tool. The attributional assessment of fuel production emissions allows California to incentivise efficiency improvements at individual biofuel plants, which is one goal of the policy. Including the ILUC term then encourages the supply of fuels from feedstocks believed to have lower overall ILUC impact, which is a second goal of the policy.

The hybrid result – an emission factor calculated as the sum of attributional direct emissions and consequential ILUC emissions – is not the most analytically relevant answer either to the lifecycle analysis question, "What emissions are associated with the processes required to produce a unit of biofuel by growing a given feedstock?", or to the lifecycle analysis question, "What is the expected change in net global emissions if we require the supply to the transport of an additional unit of biofuel?" This is an analytical compromise that enables us to take an attributional lifecycle analysis result and adjust it so that it provides a more useful indication of what the consequential emissions of biofuel supply might be.

The idea of the complementary use of attributional and consequential approaches is promoted by Brander et al. (2019), which proposes a two-step lifecycle accounting and decision-making process whereby attributional LCA is used to help an operator understand the local impacts of a process, and a consequential LCA is used to identify the system-wide consequences of available choices. At the regulatory level, this could be implemented by using consequential LCA to inform decisions about what level of support to offer biofuels in general from a given feedstock but requiring operators to undertake attributional LCA to allow more efficient processes to be rewarded. The California hybrid approach deals with this through the construction of a single hybrid LCA value, but the two elements can also be separated out. Implicitly, the European Union already applies attributional and consequential thinking in a complementary way by offering stronger support to advanced biofuels. While there is no single consequential LCA result that is used to justify the creation of a sub-target for advanced biofuels, the subtext is that the EU has been convinced that, even though advanced biofuels and first-generation biofuels might have the same reportable GHG performance under the RED, advanced biofuels deliver more GHG benefit across the system as a whole.

3 Economic models for assessing ILUC

As we discussed above, consequential lifecycle analysis of biofuels requires the development of scenarios for what will happen in the world in response to some given change in biofuel demand or biofuel policy. Developing these scenarios requires some sort of model to be developed of the way that decisions are made, and economic theory is an obvious place to go to find the basis for such a model. Economic models of ILUC, which can be divided into 'partial' and 'general' models depending on whether they address only the most relevant sectors (e.g. agriculture) or the whole economy, have become the main tool used by policy makers attempting to quantify potential ILUC emissions. In this section we discuss those models in more detail, and then in section 3.5 we discuss some alternative or complementary approaches that have been developed.

An economic modelling approach involves adopting an underlying framework in which we treat people and businesses as making decisions that are rational in terms of financial outcomes. If a farming business can make better returns by growing more wheat and less rapeseed, then in an economic model that is what it will do. In an economic model, information is exchanged through prices, and economic actors react to changing prices. This is clearly a simplification of real-world decision making – individual actors may make decisions for a range of motivations some of which are directly related to prices ("the price of corn has gone up, this season I will plant more corn"), some of which are indirectly related to prices ("I read an article that said that an ethanol mandate is going to be introduced and I think that it will cause corn prices to go up, so this season I will plant more corn") and some of which may be unrelated to prices ("I'm tired of being an investment banker, I'm going to buy some land in the countryside and become a corn farmer"). The implied assumption in economic modelling is that, when considering the economy as a whole, decisions can be usefully modelled as price led and financially rational even if in reality motivations are more complex.

3.1 How economic models work

An operative economic model consists of a set of mathematical rules that govern the responses of economic actors to changing prices. The models used for ILUC modelling are often described as general or partial equilibrium models. Equilibrium modelling involves constructing a baseline that is in economic equilibrium (supply matches demand and prices are in balance so that no industry is making losses or excess profits). Some exogenous change[13], sometimes referred to as a 'shock', is then made to one or more numerical values in the model (for example increasing biofuel demand) and the model is allowed to find a new balance, generally through a computationally intense iterative process. Figure 3 provides a much-simplified schematic representation of the sort of adjustments that occur as an equilibrium model looks for a new equilibrium following a shock.

[13] Exogenous means originating outside of the model structure, i.e. a change made directly by the modeller. This is contrasted with endogenous changes, which emerge from the numerical rules of the model.

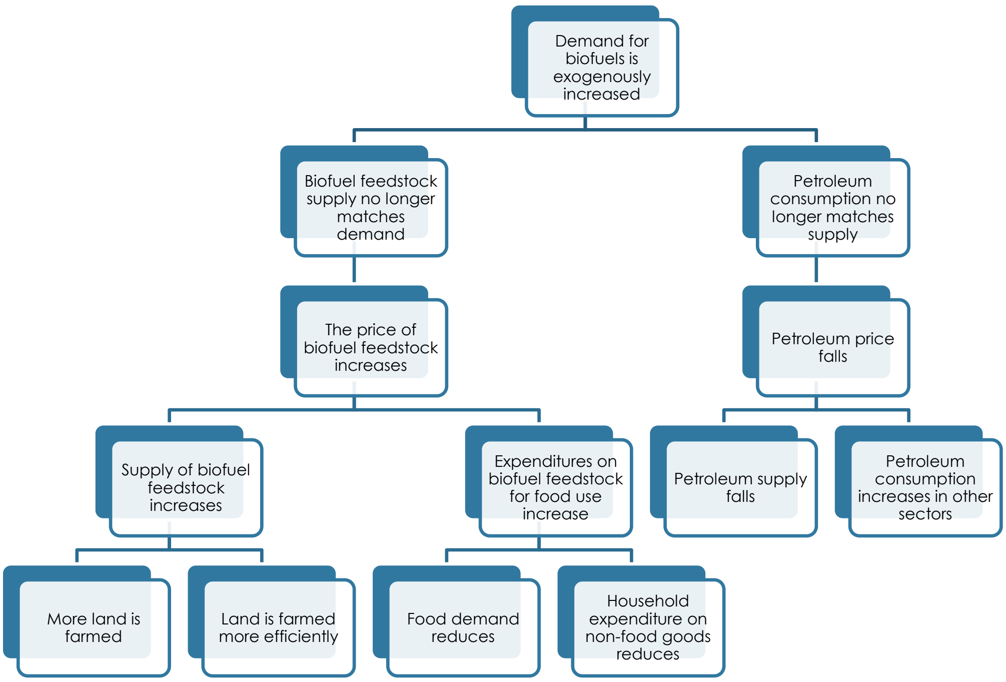

Figure 3 Schematic representation of the cascade of adjustments when finding a new model equilibrium

Depending on the level of regional disaggregation present in a model, a shock could be applied at the national level (as in modelling of corn ethanol mandates in the U.S.) or regional level (for instance the EU). The location of a shock can be important because land use change dynamics can differ between regions. In the case of soybean production, for example, while soy is grown in both North and South America, most soy-associated deforestation occurs in South America. We might therefore expect that there will be differences in land use change emissions outcomes for the creation of new soy demand in the United States versus Argentina, even if the markets are somewhat connected.

While the basic premise of adjusting to find new equilibria in response to shocks is the same for all equilibrium models, the mechanisms the models use to decide how to reach a new equilibrium differ considerably. The most fundamental difference is between 'partial' and 'general' equilibrium models. Partial equilibrium models include only a few sectors of the economy but do so in considerable detail, whereas general models include the whole economy but do so in less detail.

Partial equilibrium models tend to work in terms of physical quantities such as tonnes of material produced and area of land used, whereas general equilibrium models tend to work in more financial terms, for example considering land in terms of rents rather than physical areas. Prices allow physical units to be converted to and from financial units, but it is noteworthy that using financial flows as a functional unit in general equilibrium modelling can lead to counter-intuitive outcomes – for example, if explicit constraints aren't introduced in general equilibrium models, then quantities like mass or area may not be preserved.

Partial and general equilibrium modelling are not mutually exclusive philosophies, and it is possible in principle to use the results of one approach to inform the other. For example, partial equilibrium modelling tools that have more detailed characterisations of agricultural decision making may be used to calibrate the agricultural sector responses in a general equilibrium model. Similarly, results relating to the broader economy from a general equilibrium model (for example relating to the fossil fuel rebound) could be used to apply adjustments to agricultural results from partial equilibrium modelling. One exercise coupling general and partial modelling frameworks is presented by Britz & Hertel (2011), which couples a partial-general modelling framework using the CAPRI (Common Agricultural Policy Regionalised Impact) model for the agricultural sector in Europe and the GTAP (Global Trade Analysis Project) model for the rest of the global economy. In this work, the EU agricultural market behaviours predicted by CAPRI are mathematically integrated into the broader GTAP framework. This is shown to result in significant differences in the location of predicted production and land use changes, and allows more detailed results to be reported within the EU.

While Britz & Hertel (2011) firmly endorse the idea of coupling models in this way, the practice has not been widely adopted in ILUC analysis. This may be explained in large part by the challenges associated with coordinating work between different modellers and modelling teams. The economic modelling tools used for ILUC modelling were originally developed to assess other questions, generally relating to analysis of agricultural policy and of the impacts of changing trade rules, and continue to be used for these purposes. Despite its importance in biofuel policy, ILUC analysis is not the only or even main concern of the communities that develop the various economic models, and the resources available for ILUC modelling are limited. In the EU, ILUC modelling has been undertaken under contract to the European Commission, and bids must be competitive if they are to succeed. The overheads associated with coupling different modelling frameworks together are significant, and therefore unless contracting authorities make it an explicit requirement, modellers are likely to prioritise the development of a single modelling system over the integration of separate systems.

The details and mathematics of economic model structure are necessarily complex, and therefore many authors have tried to simplify the explanation of ILUC modelling by focusing on the most important underlying factors that dictate the amount of land use change predicted and the associated GHG emissions. Woltjer et al. (2017) presents a decomposition approach that splits ILUC modelling into six basic steps:

1. Identify the gross land demand, i.e. the amount of land necessary to grow the amount of feedstock required for biofuels at typical yields;
2. Area requirement is reduced if co-products are returned to the market;
3. Area requirement is reduced by reductions in demand for agricultural commodities for other uses (primarily food, feed, pharmaceutical);
4. Area requirement is reduced if yields of crop production are increased;
5. Area requirement is affected (up or down) by relocation of crop production to areas with different yield potential;
6. GHG emissions are assessed based on the land use changes predicted.[14]

In practice, economic models address all of these steps at once, and the net changes the system are an aggregate across thousands of productivity shifts, demand changes and production changes across the modelled system. Identifying broad themes is useful

[14] We note that the decomposition as presented in (Woltjer et al., 2017) relates to land use area only, we have added to the list the step of conversion of area changes to emissions.

because it provides a more narrative way to engage with the model outcomes, and to compare modelled behaviours with observed behaviours. For example, when considering the second step (impact of co-products) we can look at evidence from livestock feed markets and from analysis of the properties of different feeds, and then ask whether the outcomes in a given model seem consistent with what our understanding of feed markets leads us to expect (this is discussed further in section 3.4.2).

3.2 Partial equilibrium approaches

Partial equilibrium models used for indirect land use change are generally limited in their sectoral coverage to the agricultural and/or forestry sectors (Marshall et al., 2011). The advantage of being sectorally limited is that partial equilibrium models are able to provide a detailed characterisation of agricultural systems (Clark, 2018). In particular, partial equilibrium models are able to include explicit characterisation of agricultural decisions such as labour use, fertiliser use, crop rotations, and livestock management systems. Some partial equilibrium models are able to include an explicit spatial characterisation of land use – for example the GLOBIOM model divides the world into over 10,000 productive 'units' (Valin et al., 2015).

Because partial equilibrium models only cover some sectors, they do not allow for adjustments elsewhere in the economy. For example, increasing biofuel mandates will affect the local price of transport fuels. Locally, an increase to a biofuel mandate can be expected to increase retail prices of transport fuels, because biofuels are more expensive than petroleum fuels[15]. Globally, however, we expect the price of crude oil to fall slightly if petroleum demand is marginally reduced. Changed fossil fuel prices could in turn affect consumer demand – if at the global level consumers save money on fossil fuel purchases this would allow them to purchase other goods, some of which could have an associated land use burden or other associated emissions.

In general commentators have concluded that this limitation of scope in partial models should not be seen as a major drawback in the context of land use change modelling. Some analysts and commentators may argue that as a model moves further from the agricultural economy the already considerable uncertainty in ILUC modelling only increases, and that it is inappropriate to either give credit to, or penalise biofuels for, outcomes in general equilibrium modelling that are only very tangentially linked to the bioeconomy. The counterargument would be that if indirect effects are to be assessed at all, we should model the impacts as broadly as we are able, and that it is arbitrary to assess indirect effects only in the agricultural economy when we might expect further land use impacts mediated through other sectors.

3.2.1 GLOBIOM

In the European Union, the partial equilibrium model that is currently of most interest for ILUC modelling is the GLOBIOM (Global Biosphere Management) model. This is

[15] The impact on local fuel prices is sensitive to the type of biofuel incentives introduced. While a mandate imposes the extra costs of biofuels on fuel suppliers and therefore eventually on drivers, if incentives are given by reduced fuel tax on biofuels this could support lower fuel prices by imposing costs on taxpayers generally rather than on drivers specifically.

because the GLOBIOM framework was chosen by DG Energy as the basis for the European Commission's most recent ILUC emissions estimates (Biggs et al., 2016; Valin et al., 2015). In this sense, GLOBIOM succeeded the general equilibrium model MIRAGE, which has been used in previous studies, as the European Commission's primary ILUC modelling tool. GLOBIOM modelling is led by the International Institute for Applied Systems Analysis (IIASA).

The choice of GLOBIOM for this modelling work was based on the results of a call for tenders by the Commission, and the qualities of the GLOBIOM modelling framework will have been one of a number of factors assessed by the Commission when choosing between bids. To the best of our knowledge the European Commission has not publicly stated whether bids were received for this work proposing to use other modelling systems, or stated the reasons for choosing GLOBIOM over other options. One should therefore be cautious of interpreting the choice of the DG Energy to work with GLOBIOM for these studies as an endorsement of partial equilibrium modelling frameworks as such.

The ILUC estimates from Valin et al. (2015) are shown in Figure 4. Like earlier modelling with MIRAGE (discussed in section 3.3.2) the results suggest a hierarchy with vegetable oil crops being associated with more ILUC emissions than cereal or sugar crops. Valin et al. (2015) is also notable for modelling very high ILUC emissions for palm oil and (to a lesser extent) soy oil, several times greater than the other vegetable oils considered. This distinguishes these results from the results presented in Laborde (2011), which were relatively similar across vegetable oils.

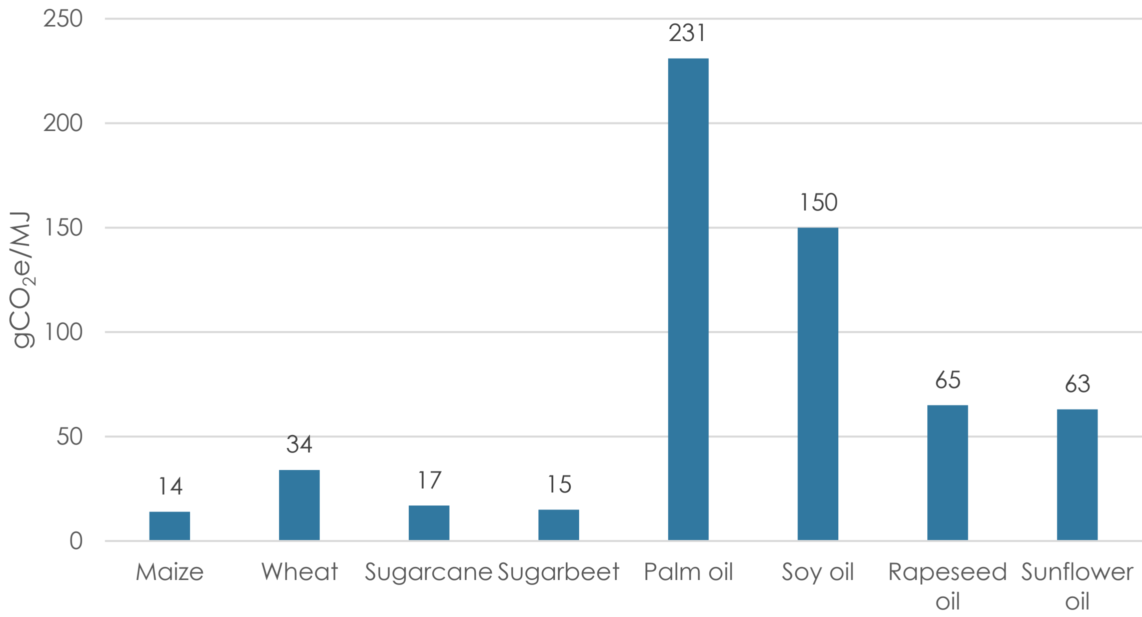

Figure 4 ILUC results obtained with GLOBIOM for the EU

Source: Valin et al. (2015)

GLOBIOM has also been used more recently in the process of developing ILUC values for use in the International Civil Aviation Organisation's CORSIA system for emissions offsetting (cf. ICCT, 2017). The scenarios modelled in the ICAO context (ICAO CAEP, 2019) have some differences from the earlier work, including that the final fuels modelled are aviation compatible (meaning slightly different energy conversion efficiency than is achieved for road fuels) and that results are modelled with increases in biofuel demand in regions other than Europe. The ICAO analytical process has also included some harmonisation in assumptions between the GLOBIOM model and the GTAP model[16] with a view to bringing the output ILUC values closer together for the two models, which has contributed to the difference between these newer ICAO results and the 2015 results (Malins, 2019b).

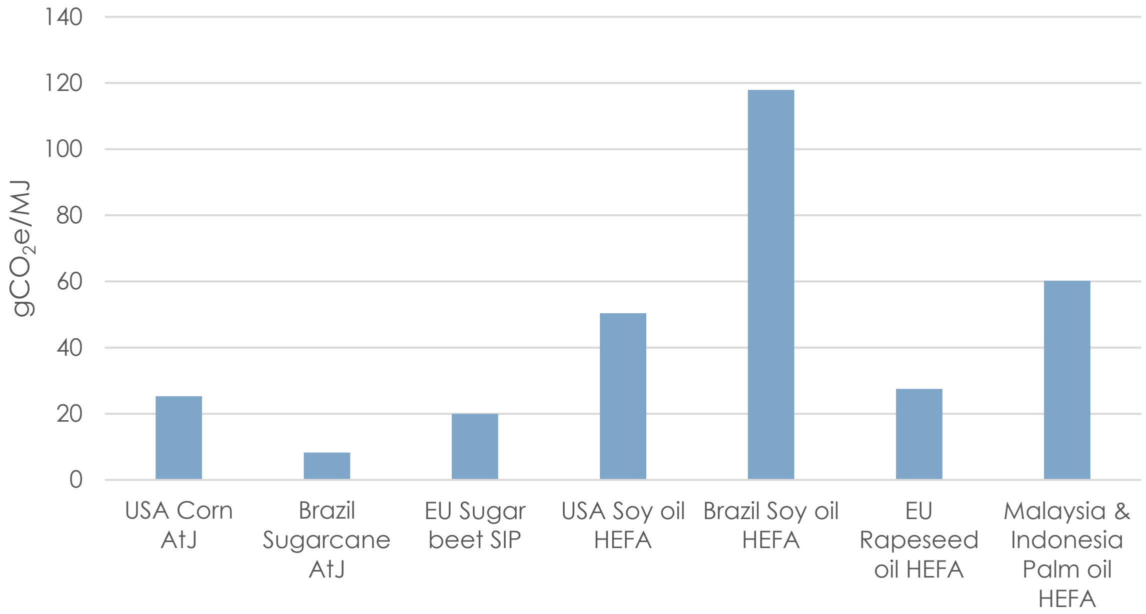

Figure 5 ILUC results obtained with GLOBIOM for CORSIA

Source: ICAO CAEP (2019)

3.2.2 FAPRI/FASOM

The U.S. EPA used two coupled partial equilibrium models, FAPRI and FASOM, in its ILUC analysis for the Renewable Fuel Standard (RFS, see section 6.2.1). FAPRI is an international model whereas FASOM considers only U.S. markets. While FAPRI includes a characterisation of the U.S. agricultural sector, FASOM was considered to offer a more detailed assessment and has the advantage that it directly included characterisation of changes in fertiliser and energy use that can be used to model changes in agricultural GHG emissions. It is noteworthy that FAPRI and FASOM were not fully integrated for this analysis, and thus there are some inconsistencies between the U.S. results from FAPRI and the FASOM results. The final ILUC values (Figure 6) combine the estimated U.S. domestic land use change result from FASOM with the land use change results from all other regions output by FAPRI (the U.S. land use change results from FAPRI are therefore not included in the final ILUC assessment).

The FAPRI-FASOM modelling was also used to provide a consequential assessment of changes in other agricultural emissions, including fertiliser emissions, livestock emissions and rice paddy emissions. For the U.S., the calculation of those emissions is endogenous to FASOM, while outside the U.S. this was done by multiplying modelled

[16] The GTAP model is introduced in section 3.3.3.

changes in crop area and livestock numbers by regional average fertiliser application rates and livestock emission factors.

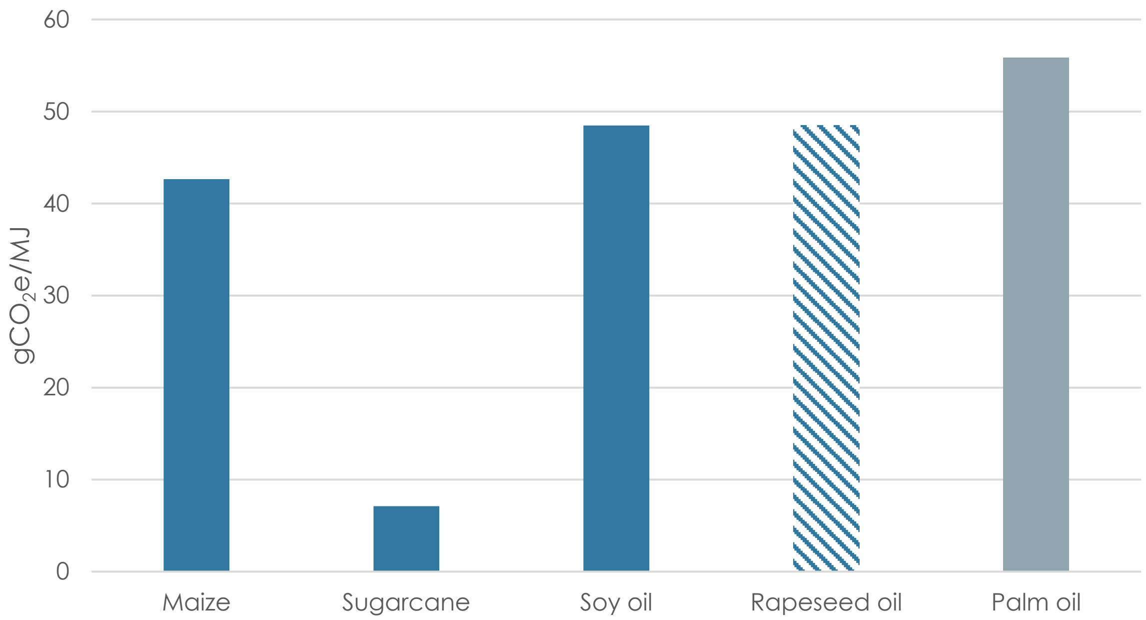

Figure 6 ILUC results obtained with FAPRI/FASOM for EPA

Source: (U.S. Environmental Protection Agency, 2010; U.S. EPA, 2011).
Note: Values adjusted from a 30 year to a 20-year amortisation. The palm oil analysis has never been finalised. Rapeseed (canola) oil is assigned the same ILUC factor as soy oil.

3.2.3 AGLINK

The AGLINK-COSIMO model was used by the JRC (European Commission Joint Research Centre) in one of the first ILUC modelling exercises for the European Commission (Blanco Fonseca et al., 2010). This study reported results including price changes, trade balances and land use changes within the EU and internationally, but it did not couple the land use change results to a land use change emissions model and therefore did not produce any outputs in terms of GHG emissions due to ILUC. The lack of ILUC emissions results from this study meant that it had much less impact in the policy discourse than contemporaneous work with MIRAGE. A subsequent JRC study (Hiederer et al., 2010) coupled AGLINK modelling results with a carbon stock change model and estimated an average ILUC factor of 63 gCO_2e/MJ for an aggregate feedstock mix to satisfy the RED targets – this work does not include estimates by feedstock. This value is about 50% higher than values reported in the same study based on MIRAGE land use change results. Blanco Fonseca et al. (2010) also included results from two other partial equilibrium models, ESIM (the European Simulation Model) and CAPRI. These models consider only impacts within the EU, and therefore on their own are not suitable for the calculation of ILUC factors given that significant ILUC emissions are expected internationally (as noted above, Britz & Hertel (2011) presented results that coupled CAPRI to an international general equilibrium model to allow international ILUC results to be calculated).

3.3 General equilibrium approaches

General equilibrium modelling differs from partial equilibrium modelling because the entire economy is included in the model, though generally in rather less detail than is possible in partial equilibrium models. General equilibrium models (often abbreviated as CGE for 'computable general equilibrium') tend to rely less on directly modelling agricultural systems, relying instead on the use of mathematical production functions defined by some functional form. The reactions to price changes in general equilibrium models are generally determined by systems of elasticity parameters, expressing the expected percentage change in some quantity in response to a given change in price. An elasticity value of 0.1 means that a given value will increase by 1% for every 10% change in the relevant price. In the process of finding a new equilibrium, thousands of quantities will be adjusted based on such elasticity-governed relationships (including elasticities between the prices of different goods, modelling for example how the price of oilseeds changes when the prize of corn increases).

The production of goods in general equilibrium models is determined by production functions. Production functions specify which inputs (sometimes referred to as 'factors of production') are required to produce a given good, and the extent to which these inputs are substitutable. For most processes, the production function requires intermediate inputs and 'value added' inputs (Figure 7). The intermediate inputs are often required in a fixed ratio, for example car manufacture may require fixed ratios of steel, rubber, plastics and glass as inputs. The classic example of value-added inputs would be land, capital and labour, but some general equilibrium models may further disaggregate these (e.g. skilled and unskilled labour), and models may include additional terms such as energy. The value-added inputs are allowed to substitute each other, for example additional labour or capital use would allow for reduced land use, which for agricultural production would be a form of productivity increase.

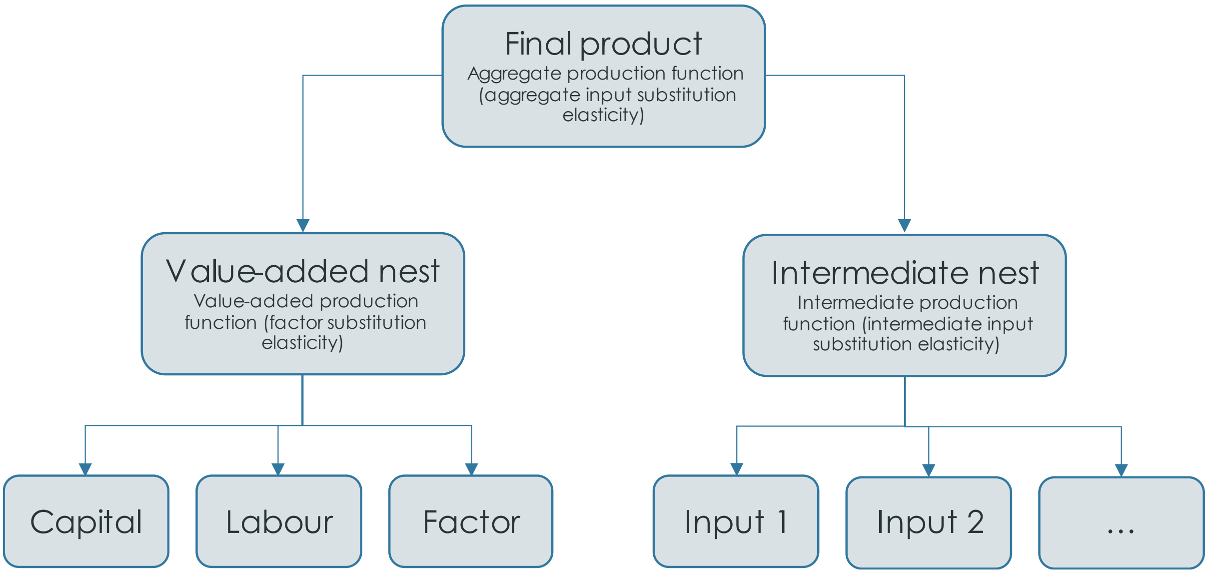

Figure 7 Schematic of a standard production function in a general equilibrium model

Source: Shutes et al. (2012)

Where a partial equilibrium model might generate an increase in productivity through explicitly adjusting agricultural choices in response to price (such as fertiliser application or choice of crop rotation), general equilibrium models would calculate a change in productivity without explicitly determining how it is achieved. The underlying economic logic is that farmers and agribusiness more generally have a variety of options they can turn to in order to increase yields (better seeds, more inputs, precision agriculture, new machinery, etc.), and that while for a given farm it may be possible to assess these options explicitly, across the system as a whole it is enough to assume that the sum of the results of hundreds of thousands of individual production decisions can be approximated with some functional form. If the elasticity of yield to price for corn is set to a value of 0.1[17] in one of these models, this is like saying that if the price of corn increases by 10%, we expect that on average farmers will increase their delivered corn yields by 1% through some combination of local decisions.

Resources and production systems in general equilibrium models are grouped together into tiered 'nests', as illustrated in Figure 8. A nest includes several quantities describing parts of the economy that can be substituted with each other to some degree, and can then be aggregated to a value which may itself be in another higher-level nest. For example, a general equilibrium model might have a cropland nest including the rents (and therefore implicitly land areas) for various different crops such as wheat, corn and rice. Including these in a single nest tells the model that land area may be exchanged between the three crops. The rents/areas of these three crops can be aggregated into an overall cropland parameter which can then be nested with other land uses such as forest land and pastureland.

The left side of Figure 8 shows the structure of land use nests in MIRAGE. At the top level[18] on the left we see that an 'all crops' nest is grouped with pasture into an 'agricultural land' nest. Within this agricultural land nest, land can move from pasture to cropland and back if the relative rents change, parameterised by an elasticity value. The potential for shifts in land use is also related to the current relative uses – where roughly equal areas are committed to different land uses in a nest, we will see land move more easily back and forth than if one land use is already dominant. 'Agricultural land' can then be thought of as an aggregate term representing both pastureland and cropland, and it is grouped at the next level down into a 'managed land' nest alongside 'managed forest'. Again, as relative rents change land can move between agricultural land and managed forest based on some elasticity value. If land shifts from agricultural land to managed forest this will be reflected at the next level up by some land being taken from pasture use and some land being taken from crop use.

[17] Note that there need not be a single input value in a given modelling framework that is called the elasticity of yield to price – it could also be a property that 'emerges' from the values of several other parameters such as the elasticity of substitution between land, capital and labour, and that must be tuned rather than directly set.

[18] Top refers only to the placement on the diagram, schematics such as this could equally well be presented the other way up, so that the top would become the bottom.

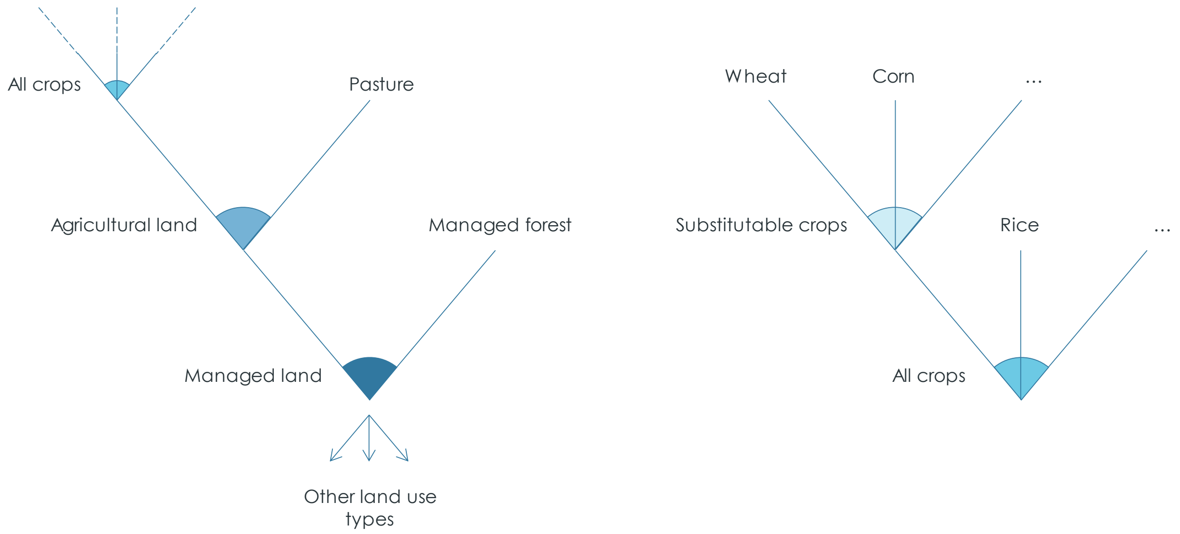

Figure 8 Examples of nesting structure from MIRAGE

Source: Valin et al. (2015)

The right of Figure 8 shows the arrangement of crops within the 'all crops' nest (which was at the top level on the left side). Notice that MIRAGE separates out rice from other 'substitutable crops'. This reflects an assumption that land will move more readily between crops with more similar management needs. It is easier to plant corn on a field that produced wheat in the previous year than to plant either of them into a former rice paddy.

General equilibrium models will normally show more substitution between resources or goods that are at the same level of the nest structure. Imagine the case that we add a wheat ethanol mandate and the rent on land used for wheat increases in our model by 20%, and assume that the only land uses present are those labelled in Figure 8 (i.e. only corn and wheat as substitutable crops etc.). At the top level on the right, we would have some reallocation of land from corn to wheat. The model would then calculate a new average rent for land used across those two crops – let's say that the split between corn and wheat in this region is roughly 50:50, in which case the rent on the 'substitutable crops' aggregate will have increased by only about 10%. At the next level down, we then allow for some shift from rice production to substitutable crop production based on this 10% rent change. The change in the aggregate rent on the 'all crops' nest may be only 6%. At the next level below that (moving to the left of the figure) we then have some transfer of land from pasture to 'all crops', and calculate the aggregate rent change for the 'agricultural land' nest which may be only 3%, and so on.

If land is split evenly between uses and the elasticities of substitution are the same for every nest, then this structure guarantees that the largest net shift in land use would be from corn to wheat, then from pasture to wheat, then from managed forest to wheat. If, however, land is not evenly split, or if some of the elasticities of substitution are lower, this hierarchy could be broken. For example, if the elasticity of substitution between substitutable crops and rice is much lower than the elasticity of substitution between pasture and all crops, or if the area of rice is very small and the area of pasture very large, the model would be able to predict a larger net shift from pasture to wheat than from rice to wheat. The framework of nests and substitutions is therefore able to model a wide range of outcomes. Running a general equilibrium model involves simultaneously adjusting resource use and consumption across thousands of such nests until reaching a new equilibrium (supply is again equal to demand).

3.3.2 MIRAGE

MIRAGE (Modelling International Relationships in Applied General Equilibrium) is a general equilibrium model that uses the GTAP database and has many features in common with the GTAP general equilibrium model. DG Trade is a member of the consortium that develops and uses MIRAGE, and commissioned MIRAGE modelling to inform European Commission decision making in the first phase of the ILUC discussion (Al-Riffai et al., 2010). Unlike other modelling results developed in this phase of the European process (see section 3.2.3), (Al-Riffai et al., 2010) presented fully global results and reported emission factor results for individual feedstocks in units of gCO_2e/MJ. Arguably it was the ability to provide results in the format most relevant to the policy discussion, rather than a more thoughtful comparison of model strengths and weaknesses between MIRAGE and the other models available, that led to updated MIRAGE results (Laborde, 2011) becoming the basis for the ILUC estimates included in the ILUC Directive. The MIRAGE modelling included two trade policy scenarios – a current policy scenario and one with elimination of biodiesel tariffs. Given that tariffs on biodiesel imports remain in place on the main potential exporter nations, we will consider only the results from the current policy scenario (they are in any case similar results for all feedstocks).

The feedstock level results from Laborde (2011) are shown in Figure 9.

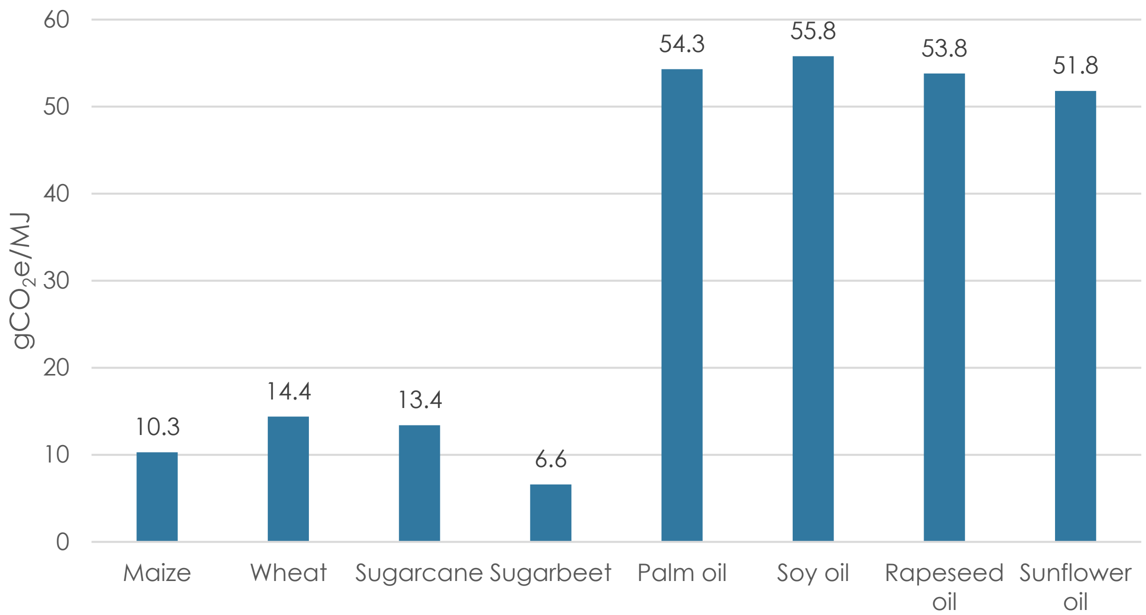

Figure 9 ILUC results obtained with MIRAGE for the EU

Source: Laborde (2011)

3.3.3 GTAP

The computable general equilibrium model of the Global Trade Analysis Project (GTAP for short) has been used for ILUC modelling by the California Air Resources Board in the LCFS, alongside GLOBIOM in analysis for CORSIA, by the Argonne National Laboratory for ILUC estimates include in its GREET model (Argonne National Laboratory, 2017) and in a series of publications authored by academics at Purdue University in the U.S. There are probably more iterations of ILUC modelling published using the GTAP model than any other model. One factor contributing to this regular stream of publications is that fact that the main GTAP modelling team is based in a university department rather than another type of institution. The ILUC values used for regulatory compliance in the California Low Carbon Fuel Standard are illustrated in Figure 10. These results are not from a single scenario but are the average emissions across thirty scenarios for each feedstock calculated for the Air Resources Board.

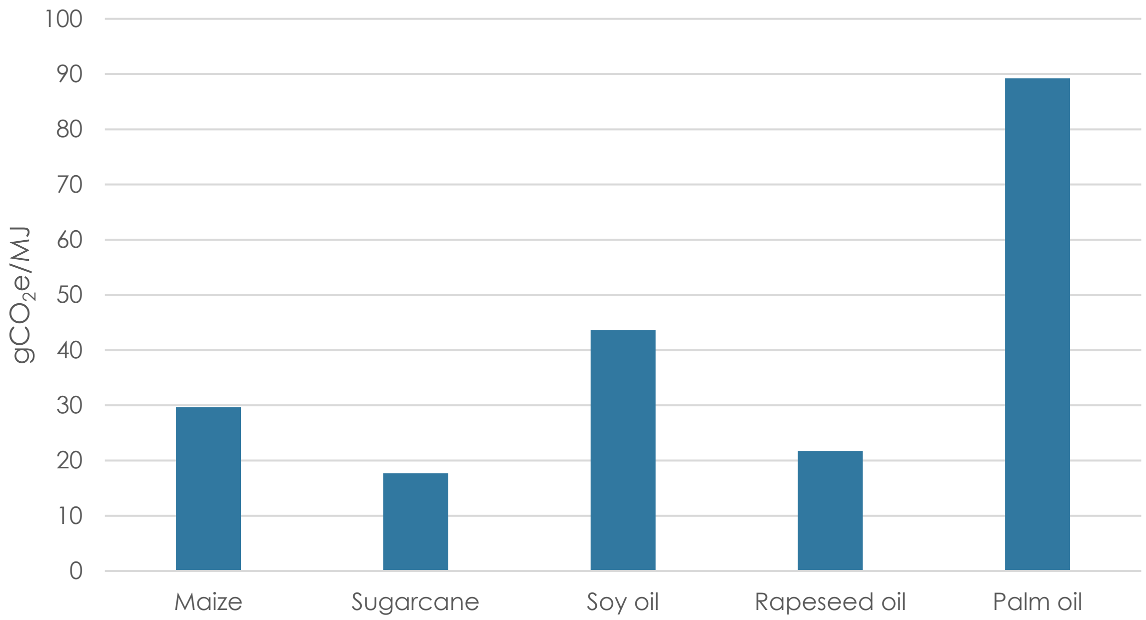

Figure 10 ILUC results obtained with GTAP for California Air Resources Board

Source: California Air Resources Board (2014)
Note: Values adjusted from a 30 year to a 20-year amortisation[19]

The hierarchy of emissions values is similar to that delivered by the MIRAGE modelling of Laborde (2011); palm oil has the highest assessed value followed by soy oil, ethanol crops have lower values. One difference is the relatively low value calculated for rapeseed oil (referred to as canola in North America). California Air Resources Board (2014) provides relatively little detail of the results of the modelling, but the land use change and land use change emissions results by region and AEZ are provided for one of the thirty scenarios as part of the package for version 52 of the AEZ-EF (agro-ecological zone emission factor) model used in the LCFS modelling (Plevin, Gibbs, et al., 2014)[20]. These results (Figure 11) show that the area of land use increase predicted is significantly larger in the soy scenario, but also that the location of expected land use changes are quite distinct. While both predict the bulk of land use changes to occur in North America, the rapeseed scenario shows expansion in Canada while the soy scenario shows expansion in the U.S. itself. The soy scenario also shows larger area increases in South America and Southeast Asia. The lower amount of modelled palm

[19] One-off emissions from land conversion are simply adjusted to the 20-year amortisation by multiplying by 1.5 (as the emissions are divided over a shorter period). Peat emissions are considered to persist over the period considered, and therefore we treat the estimated share of annualised peat emissions in the ILUC values for palm oil as independent of amortisation period.

[20] https://www.gtap.agecon.purdue.edu/resources/res_display.asp?RecordID=4346

oil expansion in the rapeseed scenario is a significant contributor to the lower resulting ILUC value.

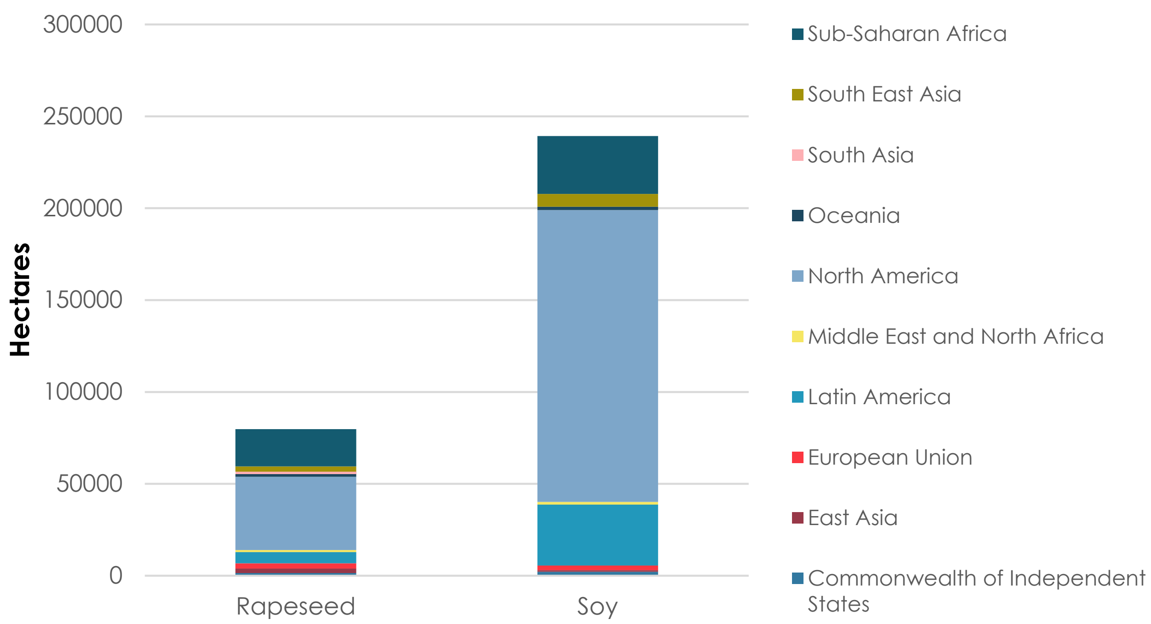

Figure 11 Location of cropland expansion for U.S. rapeseed oil and soy oil scenarios with GTAP.

Source: own calculations based on soy and canola scenarios 6 from California Air Resources Board (2014). Note that the soy result has been normalised here to match the rapeseed oil shock size of 400 million gallons.

It is also important to recognise that ILUC modelling results can be quite different depending on the region in which the demand shock is modelled. Figure 12 shows that the predictions for location of cropland expansion in response to rapeseed demand are completely different between GTAP modelling for U.S. demand and GLOBIOM modelling for EU demand. In GLOBIOM, the main locations for cropland expansion as EU demand for rapeseed biodiesel increases are within the EU itself and to a lesser extent in North America, sub-Saharan Africa and Southeast Asia. In GTAP, the biggest land use changes are in Canada and sub-Saharan Africa with relatively little expansion in Southeast Asia. At least equally importantly for the results, the GTAP modelling for U.S. demand predicts only a tenth as much net land expansion per unit of energy produced. This implies a much stronger productivity response in the GTAP modelling.

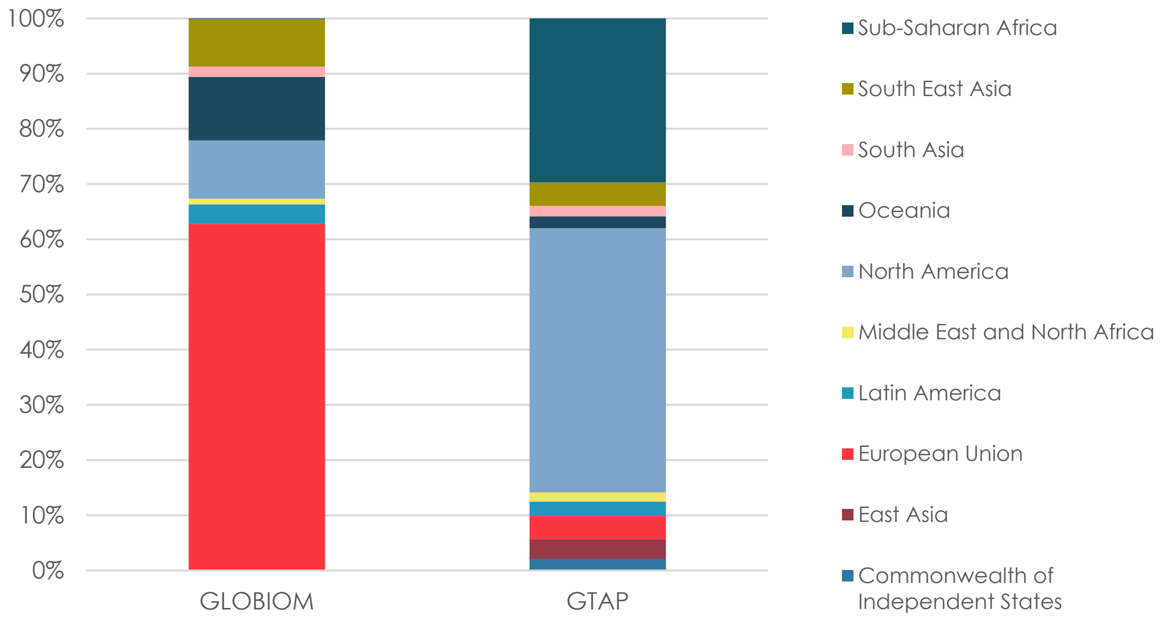

Figure 12 Location of cropland expansion for EU rapeseed oil biodiesel scenario with GLOBIOM and U.S. rapeseed oil scenario with GTAP

Source: own calculations based on canola scenario 6 from California Air Resources Board (2014) and results from Valin et al. (2015)

Like GLOBIOM, GTAP has also been used in the development of ILUC factors for CORSIA – these estimates are shown in Figure 13. GTAP modelling has been subjected to considerable scrutiny both in the context of its use in the LCFS and its development independent of the California Air Resources Board. Malins et al. (2020) raises a number of questions relating to the model, and in particular is rather critical of the strong role that agricultural intensification has been given within the GTAP framework (and the consequent reductions in modelled ILUC emissions).[21] It is suggested that the evidentiary basis for changes to the modelling framework since 2009 has sometimes been weak, and that a lower standard of evidence may have been applied before introducing changes that increase intensive responses (and therefore reduce ILUC emissions in model results) than before introducing changes that could increase modelled ILUC results.[22]

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[21] A response to (Malins et al., 2020) is provided by (Taheripour et al., 2021).

[22] One example of this is that GTAP does not include a mechanism to model the conversion of unmanaged land to agricultural use. This was identified as a limitation in the GTAP framework over a decade ago and was part of the EPA's argument for preferring the FAPRI-FASOM modelling system. The MIRAGE model has a GTAP-like structure and includes a mechanism to allow conversion of unmanaged land, but this mechanism has never been copied across, nor (to the best of our knowledge) has any alternative mechanism been suggested.

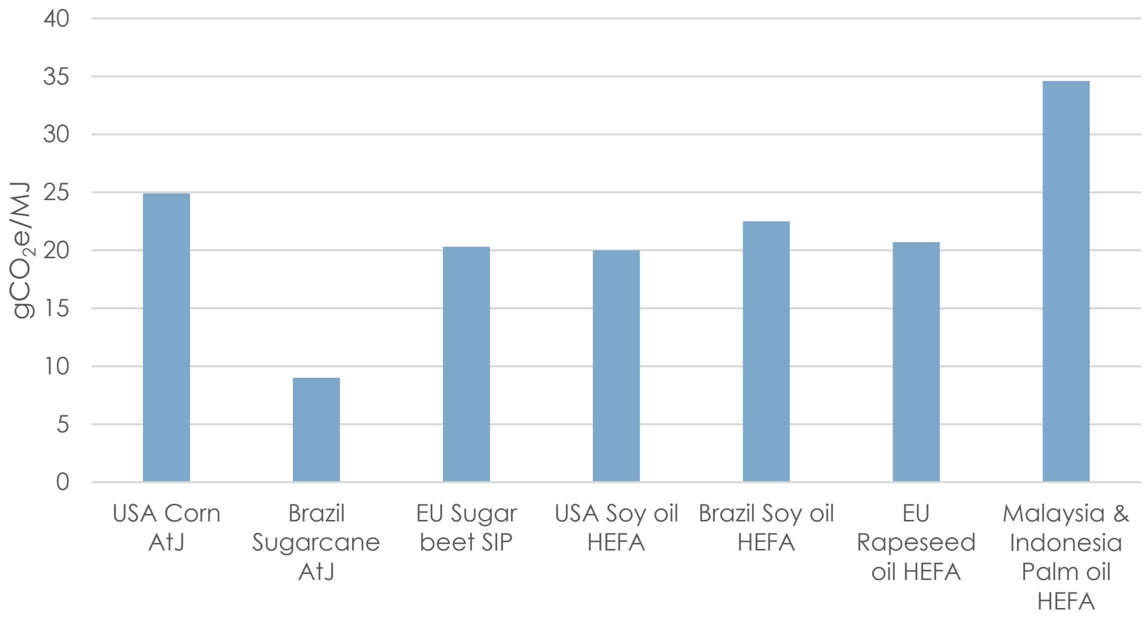

Figure 13 ILUC results obtained with GTAP for CORSIA

Source: ICAO CAEP (2019)

3.4 Partial versus general equilibrium

3.4.1 Representing agriculture

An example of the difference in representation of the agricultural sector that is possible in partial versus general equilibrium tools may be found from comparing two models previously used for ILUC studies for the European Commission – MIRAGE as used by Laborde (2011) and GLOBIOM as used by Valin et al. (2015)[23].

GLOBIOM explicitly includes 18 crops globally, with a further 8 crops modelled in the EU only, while MIRAGE explicitly models 11 crops. GLOBIOM holds the area constant for the group of crops that are not explicitly modelled, whereas MIRAGE includes aggregates of 'other' crops that are able to trade land within the model. GLOBIOM disaggregates the world into 29 regions plus all EU countries, while MIRAGE disaggregates the world into 11 regions (one of which is the EU). GLOBIOM considers land use and crop production disaggregated to over 10,000 represented grid cells, whereas MIRAGE considers aggregate crop production at the regional level divided into AEZs (agro-ecological zones). GLOBIOM is able to model 12 different agricultural GHG emissions sources, while MIRAGE considers only land use change emissions. In GLOBIOM, crop yields are identified explicitly at the level of the grid cell informed by land and climate characteristics and distinguished by crop management system, whereas in MIRAGE a single crop yield is set for each crop at the region-AEZ level.

[23] This section is based on the discussion of differences between the two models provided in Appendix I of (Valin et al., 2015).

3.4.2 Dealing with co-products

Conventional biofuel production routes are notable for the generation of co-products that can be used in livestock feed. Ethanol production from cereals like corn and wheat produces distillers' grains, while crushing oil seeds for vegetable oil produces oilseed meals. The generation of these co-products has a vital role in reducing the net resource requirements of biofuel production, as was noted in section 2.5.

Take for example the case of wheat ethanol. Processing a tonne of wheat into ethanol results in the production of about a third of a tonne of distillers' grains and solubles (DGS). DGS contain less metabolisable energy per unit mass than the original wheat, but a higher fraction of protein (DGS contain essentially all of the protein from the wheat in a third of the mass, and therefore have three times the protein concentration) (Hazzledine et al., 2011). The percentage protein content of DGS is very similar to that of rapeseed meal, though DGS contains lower amounts of key amino acids. If modelling ILUC from wheat ethanol production, the role of DGS has to be taken into account. A simple model of the impact of DGS supply would be to assume that because a third of the mass of the wheat is returned to the animal feed market, the net demand for wheat is reduced by a third. Some earlier ILUC modelling included assumptions of this sort, but this simple approach does not capture the characteristic difference between the characteristics of the feed materials. A more sophisticated approach requires consideration of the potential for DGS to replace high protein feeds, such as rapeseed meal or imported soy meal. Lywood et al. (2009) proposed an approach in which one would identify the amounts of soy meal and feed wheat that would be required to exactly match the protein and energy content in a tonne of DGS. From this analysis, one could conclude that DGS will primarily displace the use of soy meal in EU feed markets – but this approach is still a simplification, as livestock diets are formulated based on more than just crude protein and energy content, and the livestock market in the EU uses many other protein feeds which could also be displaced by DGS. Hazzledine et al. (2011) goes further by presenting a full linear programming approach to identify potential changes in cost-optimal diet formulations as the availability of DGS increases. This approach results in identifying a wider portfolio of feed ingredients whose consumption could be affected by DGS supply, but does not include economic factors – in practice, the supply of same feed ingredients may be more elastic to demand than others. There has similarly been much discussion about how corn DGS might affect the U.S. feed market, including whether a tonne of corn DGS always displaces a tonne of other materials or could displace more and whether it will displace protein feed or instead reduce the use of urea in ruminant feed as a nitrogen supplement (ruminants can produce their own protein if fed appropriate nitrogen).

In ILUC modelling, the challenge for modellers is to find an appropriate intersection between an understanding of how livestock rations are formulated and the capacity of a model to represent that feed market. Throughout the ILUC debate since 2007 there has been a common refrain from some corners of the biofuel industry claiming either that models do not consider co-products at all, or else do not consider them adequately. **The first claim is categorically incorrect!** Co-products have been a feature of ILUC models since the publication of Searchinger et al. (2008). We are not aware of any ILUC result produced for the European Commission in which co-product supply was not represented, and to the best of our knowledge consideration of co-products is a standard feature of all partial and general equilibrium models currently used to model ILUC emissions.

The second claim is more interesting – how well does a model need to model the livestock feed market in order to produce useful results? Looking again at GLOBIOM and MIRAGE as examples (Valin et al., 2015), we can see some of the challenges this poses in partial and general frameworks. GLOBIOM as a partial equilibrium model has a rather more detailed approach to the livestock industry. It includes a feed digestibility model representing the differing feed value of materials to different types of animal, and each livestock management system allowed by the model is associated with its own feedstock mix. GLOBIOM allows oilseed meals to inter-substitute in feed rations subject to energy and protein content matching, or for an increase in the availability of oilseed meals to result in the adoption of higher-protein feed systems. DGS are modelled in a slightly simpler way, with the model assuming that any available DGS will directly substitute other feed ingredients (soy meal and a standard local cereal feed) in accordance with defined substitution ratios determined based on protein and energy content – this is similar to the calculations suggested in Lywood et al. (2009) but is adjusted for relative digestibility of each ingredient for each animal group. This means that for some animals a tonne of DGS displaces more than a tonne of other feed, and for some animals less than a tonne. The model does not consider more detailed feed specification issues such as amino acid profiles, which may result in an over-prediction of the amount of soy meal that can be displaced. Soybeans have lower mass yield per hectare than cereal crops, and soybean production is more associated with deforestation than cereal production (cf. Malins, 2020), and therefore using strong assumptions about the amount of soy meal displaced and therefore reducing required soy area might be expected to tend to reduce ILUC estimates.

MIRAGE, as a general equilibrium model, has much less resolution in the livestock industry and does not consider details like differential digestibility directly. In MIRAGE the protein content of DGS is addressed by creating a nest of protein feeds within the livestock feed sector, which includes distillers' grains and oilseed meals. These materials are able to substitute each other, and then the protein feed aggregate may substitute other feeds (so that if average protein prices reduce, overall protein use will tend to increase). Malins (2011) notes that the substitution ratios that are reported as emerging from the MIRAGE modelling in Laborde (2011) show levels of protein feed substitution that are similar to those reported in other literature (e.g. Hazzledine et al., 2011) but lower than the soy meal substitutions assumed in GLOBIOM. MIRAGE may underestimate cereal feed substitution, reporting overall feed substitution rates less than one tonne per tonne. It is unclear whether this is because there is some additional replacement of pastureland as a source of livestock nutrition. On this basis, it seems that the co-product treatment in MIRAGE may be less favourable to biofuels than the treatment in GLOBIOM. Another interesting point noted by Malins (2011) is that the overall MIRAGE results seem to suggest that the direct substitution of soy meal is offset by a protein feed rebound – soy meal use as a share of total protein feed is reduced, but total protein feed consumption increases as a result of extra supply and lower prices. This illustrates the point, noted above, that assessments of substitution based only on nutritional value without considering economic factors may not capture the full picture of potential net changes in consumption.

Including livestock feed markets in either type of model is complex and requires simplifications. A considerable effort has been put into improving co-product handling in equilibrium models as a result of the interest in ILUC modelling, and approaches have evolved considerably since 2008, in particular by making consideration of protein content in animal feed a standard model feature. As with any aspect of the models, there remains space for discussion about whether the direct substitution assumptions are consistent with real world practices, and whether the aggregate results are plausible and consistent with observed changes in aggregate feed use, but it would be factually incorrect to claim that the nutritional profile of co-products is not addressed in the modelling.

3.4.3 Trade

Trade flows are an important part of equilibrium modelling, as agricultural demand changes affect not only the agricultural system and land use in the country where they occur but can also result in changed to imports and exports, causing land use changes further afield. Treatments of trade can be quite different in different models, and the choice of trade system does not necessarily flow from whether a model is partial or general equilibrium. Some models use a 'single world market' model, in which it is assumed that there is a world market price for imports/exports and that goods may be traded via this single world market between any combinations of countries, generally subject to some consideration of transport costs and tariff barriers. The GLOBIOM model implements a version of this approach, treating products from each region as identical and assuming that products for import will be sourced from the exporter with the lowest production costs, accounting for transport costs and tariffs and calibrated to baseline reported trade flows. An increasing cost of trade prevents the case that all trade is provided by the same region. A single world price approach allows dynamic changes in trade patterns to emerge from modelling, but it may fail to adequately account for features of global markets that are not fully captured through price information (for example the ideas that there may be market inertia favouring the preservation of existing trade relationships).

An alternative approach often used in general equilibrium models is through the implementation of 'Armington elasticities'. The Armington approach treats similar materials from different countries as being separate goods in the modelling, but which may substitute each other on the basis of bilateral elasticities of substitution that are tuned to existing trade flows. The Armington approach allows for regional preferences in trade to be established and for differential pricing between similar goods from different regions – for example, Armington elasticities might be used operationalise a preference of American consumers for Italian leather. One drawback of the Armington approach is that is does not allow for fundamental changes in trade patterns – if there is no trade relationship in a given good between two countries in the baseline, such a relationship can never develop. This makes Armington approaches potentially inappropriate if modelling the development of markets for biofuel feedstocks that have not traditionally been traded or modelling the development of new biofuel crops in regions that have not previously produced them.

Trade dynamics are important in ILUC modelling because different regions have different profiles in terms of typical yields and of expected land use change emissions from increase in agricultural production. For example, oil palm expansion in Malaysia and Indonesia is associated with the specific problem of peat drainage for land expansion, so assuming a larger contribution by those countries to meeting increased demand for vegetable oil would tend to result in higher ILUC estimates. One of the more convincing criticisms that was levelled at Searchinger et al. (2008) was that the global pattern of land use changes it predicted with a single world market approach was not consistent with expectations of which countries are more likely to respond to increased export demand, predicting too large a share of the supply response in Africa and not enough in more export-oriented markets.

3.4.4 From land use change to emissions

Equilibrium models generate results in terms of areas of land use change. Most models explicitly distinguish between types of land for expansion (explicitly including, at a minimum, categories of cropland, grassland/pastureland, and forest land). Turning area land use estimates into ILUC factors requires combining them with values for carbon stock change per area of land converted to agricultural use – this is true of both partial and general equilibrium modelling. The carbon stock change assumptions of any model are largely independent of the land use change modelling itself, and thus it is possible in principle to combine the results of one land use change modelling exercises with different carbon stock models and compare the results (see e.g. Dunn et al., 2017; Marelli, Ramos, et al., 2011).

Partial models tend to offer more spatial resolution in land use change predictions, which allows a correspondingly greater resolution in identifying expected carbon stocks. General models tend to have a coarser spatial resolution and must therefore use more aggregated carbon stock assessments. Both GLOBIOM and MIRAGE consider above and below ground carbon stock changes based on carbon stock assessments that follow IPCC assessment protocols, with additional literature review to identify appropriate estimates for carbon dioxide emissions from peat decomposition in Southeast Asia.

3.5 Partial or general?

Given that both partial and general equilibrium models are available to assess ILUC emissions, it is inevitable that policy makers and other stakeholders would want to ask whether one framework is preferable to the other. The fundamental difference between partial and general equilibrium frameworks is one of scope, though springing from this difference of scope is a difference in the type of mathematical equations that are used to represent the economy.

In terms of the impact of the scope on the results, all other things being equal we might expect that general equilibrium models would tend to return slightly lower ILUC results than partial equilibrium models. This is because general equilibrium models give the economy more ways to respond to the demand shock from a new biofuel target, including through changes in land demand mediated through non-agricultural sectors (for example an increase in agricultural land rents could cause a slight reduction in the modelled rate of urban expansion). We should also note that modelling slightly lower ILUC impacts would not necessarily mean that general equilibrium models predict greater overall climate benefits from biofuel policy. As discussed below, in section 3.6, general equilibrium models may also predict emissions increases from fossil fuel use outside the agricultural sector, although no current biofuel support policies directly consider such emissions.

While the ability to model land use reductions delivered outside the agricultural economy is a clear different between partial and general equilibrium models, this may be a second-tier concern for ILUC modellers. Land use effects elsewhere in the model are likely to be much smaller than those in the agricultural sector, and we are not aware of any general equilibrium modelling exercise having presented a decomposition to explicitly identify land use savings made outside the agricultural economy. More important to any attempt to compare the usefulness of the different modelling structures are the differences that arise between the use of generalised production functions in general equilibrium models as compared to more explicit modelling of agricultural systems in partial equilibrium models.

To put it another way, partial equilibrium models are able to explicitly consider what one might think of as agronomic information whereas general equilibrium models tend to deal in more purely economic information. Consider a simplified example in the livestock sector. A partial equilibrium model may explicitly include an intensive feedlot model and an extensive pasture model of livestock production. When prices change in the model, for instance if land rents increase faster than feed costs, the partial model may calculate that the feedlot system becomes more competitive and will show some increase in feedlot-based livestock rearing. In a general equilibrium model, we may instead have an aggregate livestock industry represented in a region with some combination of land and feed inputs that are treated as substitutable. If land rents rise faster than feed costs, the model will substitute some use of feed for use of land, but it would not directly assess the economics of the two different production systems. The partial equilibrium version generates predictions that might be considered more concrete and transparent, and therefore may be easier to compare to real life observations and experience.

The general approach is not without appeal, however. It could be argued that when a partial model adds explicit representation of production systems this is a limitation. Farms can move between defined systems, but cannot do anything that the modellers haven't foreseen. There is a sense in which the higher-level approach of the general models is less restrictive because it does not impose assumptions about ***how*** farmers change their output and use of inputs. Rather, the economic logic of the general equilibrium model assumes that, at the aggregate level, farmers are able to do *something* do improve their productivity when prices change.

With a longer-term view, partial equilibrium models might be seen as having more potential to be developed to deal with detailed agricultural market behaviours. Whereas general equilibrium models tend to be limited to using the same functional forms for all production functions and substitutions, partial equilibrium modelling offers more flexibility to model real processes. Ever increasing computational power offers the potential to increase the level of complexity in both partial and general equilibrium modelling – for instance by adding more detail about the reality of production systems in general equilibrium models, or by adding relevant sectors to partial equilibrium approaches.

3.6 The fossil fuel rebound

Above, we noted that general equilibrium modelling allows for land use demand outside the agricultural economy to be considered. It also allows in principle for energy-related emission changes to be assessed (this is also possible with a partial equilibrium model if it is extended to include the energy sector as well as the agricultural sector).

The basic principle of the introduction of biofuel mandates is that by increasing the supply of renewable fuels you reduce the consumption of fossil fuels by an equal amount, but if we look at this proposition from an economic viewpoint it becomes clear that it is not necessarily true. Economics tells us that, in general, reducing demand for a given good will tend to reduce its price, and that reducing the price of a given good will tend to create new demand. If we reduce demand for crude oil in the market with a biofuel mandate, this tends to reduce the global oil price, and consumers in other markets are expected to consume more oil as a result – total oil consumption rebounds. Such rebound effects have been studied in the context of efficiency improvements and the context of biofuel mandates. There is no more consensus on the expected magnitude of fossil fuel rebound effects than there is on the expected magnitude of indirect land use change, and the oil market has some peculiarities such as the role of OPEC (the Organization of the Petroleum Exporting Countries) and the politicisation of energy security. These could distort decision making in ways that are difficult to model with standard economic principles. Some studies have suggested that rebound effects could almost eliminate the fossil fuel savings delivered by biofuel mandates (e.g. Hochman et al., 2010 presented results in which introduction of a 1.8% global biofuel blend led to a 1.6% increase in overall global fuel consumption), though other studies predict a more moderate impact. The fuel rebound is also sensitive to the way in which a policy is introduced – policies such as carbon taxes that work by making fossil fuels more expensive to consumers in some markets are expected to have less of a rebound than policies that work by making fuel less expensive to consumers (such as tax credits for biofuel production), with fuel supply mandates falling in the middle. Malins et al. (2015) report a range of estimates of the size of the rebound from 0% to 90%.

3.7 Convergence

In general, when analysts put great effort into modelling a given question in a given system (in this case the impact of biofuel demand in the agricultural system), developing several tools in parallel and gradually improving the input data and modelled relationships in all of them, we would hope to see the that the results from those models would start to converge over time, giving us a sense of confidence that we were approaching the 'true' answer. In ILUC modelling, however, convergence has been elusive. Daioglou et al. (2020) note that ten years of ILUC modelling has not eliminated the considerable ranges seen in published results, nor the uncertainty ranges reported by individual studies.

In part, this lack of clear convergence reflects the fundamental questions that remain to be resolved in ILUC modelling. As is discussed in the rest of this report, different ILUC models take completely different approaches to fundamental questions such as modelling the development of trade patterns. While different models cannot agree on issues such as in which country land use expansion is most likely to occur it should be no surprise that they differ when assessing the associated carbon costs. Indeed, in some cases apparent convergence may disguise fundamental underlying differences. Woltjer et al. (2017) comments that,

> *"Studies that show similar levels of ILUC GHG emissions may in fact not imply result robustness. This is because the studies may be displaying completely different situations, arising from differences in parametrization, regional coverage, (potential) land use changes and scenario assumptions."*

While the wide range in reported ILUC values makes the job of the policy maker more difficult, it should not be entirely surprising that such a range exists. For one, now that levels of support for biofuels have become tied to ILUC modelling either directly (through ILUC factors) or implicitly (through the effect of reported ILUC values on the willingness of politicians to extend subsidies) the biofuel industry and its supporters have an obvious interest in arguing for and producing low ILUC estimates. While there may be cases in the literature of low ILUC values reported in poor faith or based on entirely unconvincing assumptions, less direct forms of bias are also at work. There is simple cognitive bias. It is hardly surprising if it is hard to convince biofuel producers who believe they have environmentally friendly businesses that they may be part of the problem rather than part of the solution. Equally, if you talk to a dozen agricultural analysts about land demand, intensification and extensification you will find a range of sometimes contradictory views expressed. There is a subjective element to understanding any complex system, and the subjective opinions of analysts inform the way that modelling tools are built and used. If you meet an analyst who is bullish about the potential to deliver sustainable intensification and close yield gaps, you should not be surprised if a model built by that analyst predicts that intensification will be delivered sustainable and that yield gaps will be closed. If you meet an analyst who is concerned about the capacity of the world to satisfy food demand in 2050, you should not be surprised if a model built by that analyst predicts that competition between food and fuel will make it harder to eliminate hunger.

There is also a form of attention bias whereby focus and funding are not evenly distributed. Imagine you were to identify two weaknesses in an ILUC model: weakness A is considered likely to cause ILUC estimates to be too high; weakness B is considered likely to cause ILUC estimates to be too low. You could safely predict that the biofuel industry would be more likely to fund research to deal with weakness A. Even if every change made to a model makes it a better description of reality, if you only make changes that result in lower numbers the sum of these improvements could end up making the model less reliable overall.[24] On the other side of the argument, environmental groups are much more interested in finding and documenting cases in which agricultural expansion drives deforestation than in finding cases where it demonstrably doesn't.

[24] As an analogy, consider a car with two flat tyres, one on the left and one on the right. Inflating only the tyre on the left is obviously an improvement, but may still make the car harder to steer in a straight line.

4 Other models for assessing ILUC

4.1 Causal descriptive approaches to consequential modelling

Computational equilibrium modelling is by its nature complex, and the complexity of the models means that in practice they can be seen as something of a 'black box', in which the linkage from input to output is obscured by the complex relationships in the middle. In some cases this sense of a black box is heightened because the inner workings of the model are not publicly documented, but even where source code for parts of a model is available (as for GTAP) it is extremely difficult even for informed observers to come to clear conclusions about precisely why a certain set of outcomes is modelled for a given set of inputs.

"Causal descriptive" modelling (Baral & Malins, 2016; Bauen et al., 2010) responds to this sense of obscurity through complexity by reducing ILUC modelling to a reduced set of parameter choices for which assumptions can be more clearly discussed and documented. Causal descriptive modelling has also been referred to as "participative modelling" by Rosa et al. (2014), because it was conceived in part to allow a more direct link from expert and stakeholder input to reported outcomes. Bauen et al. (2010) maps out a procedure in which a review of trends in recent historical data are coupled with information from literature review and expert input to allow a scenario to be developed for the main market responses to an increase in biofuel feedstock demand.

Whereas equilibrium modelling tends to show large responses for a few commodities and markets alongside minor responses across dozens of markets, the causal descriptive approach identifies just a few crops and countries in which the largest impacts might be expected. For palm oil, for example, equilibrium models will allow for supply responses in South America and Africa as well as in Asia, but for a causal descriptive exercise we might identify Malaysia and Indonesia as the countries where responses are considered most likely and give no further consideration to responses in other countries. Similarly, a causal descriptive model might deal with co-products by considering displacement ratios as are used in GLOBIOM or MIRAGE (see section 3.4.2), but where an equilibrium model allows the impacts of co-product availability to ripple through the agricultural system, the causal descriptive approach would consider substitution of only a couple of specific feed materials in a couple of countries.

By restricting the number of relationships considered, causal descriptive modelling can add a type of transparency that is elusive in equilibrium modelling where it can often be difficult to parse how the demand increase in the region with a biofuel mandate links to specific agricultural shifts modelled in other regions. Perhaps the major drawbacks of the causal descriptive approach are that it is not well suited to considering economic feedbacks, and that it is very hard to make the decisions in a causal descriptive framework seem objective, given that experts and stakeholders are likely to disagree on key facts. One potential role for causal descriptive modelling is as a way to develop parallel scenarios to echo results from equilibrium models, as a way to make the outcomes in equilibrium models more accessible. If it is not possible to configure a causal descriptive model in a way that seems plausible and generates comparable outcomes to an equilibrium modelling run, this could indicate that the equilibrium framework needs further calibration.

4.2 'Empirical' approaches

Woltjer et al. (2017) identifies a category of studies that they refer to as empirical approaches to estimating indirect land use change. Empirical studies are characterised by the attempt to estimate ILUC impacts based on consideration of historical information about land use changes. These empirical approaches have an attributional character, in that they generally seek to find a way to attribute recorded land use changes to agricultural demand as a driver, although the details of how this is done in different studies can be quite distinct.

Approaches like this which are tied to historical data are useful in that they provide another basis to establish the order of magnitude of potential ILUC emissions, but they tend to be limited by the lack of a compelling basis to establish a causal link between expanded demand for a particular category of biofuel and conversion of particular land areas. Consider for example the work of Fritsche et al. (2010). This study is part of a line of work that is credited with coining the term "ILUC factor", although subsequently the term has become widely associated with the use of ILUC emissions estimates from economic models. Fritsche et al. (2010) develop a model for the average CO_2 cost of land use based on an underlying assumption that future commodity expansion will echo current distributions of commodity production for export. This is expressed as a value in units of tonnes of CO_2 per hectare per year, which is intended to reflect the likely average CO_2 emission causes by displacing production into the biofuel market. This ILUC factor approach is represented as a 'deterministic' approach that allows analysts to bypass the complexity of equilibrium modelling.

While there is no question that the calculation is indeed simplified compared to equilibrium modelling, the resulting critique of the Fritsche et al. (2010) approach is that all information about actual linkages between a given crop and carbon stock changes are lost – the final ILUC factor attributed to crops grown on existing farmland is derived only from the global average value per unit of land use increase and the yield of the crop in question. This means, for example, that while deforestation in Indonesia is identified as a main source of land use change emissions in calculating the global average ILUC factor, the value attributed to palm oil is about half that attributed to rapeseed oil because palm oil has a higher yield than rapeseed oil. This result is simply not convincing, not least because assuming that the same per-hectare ILUC emissions can be associated with all biofuel production is entirely arbitrary. This is an example of a general risk in such empirical approaches that in the drive to simplify results and tie them transparently to real world data the outcomes can become meaningless as a proxy for to identify the likely ILUC impacts of increases in demand for any particular biofuel, which as was discussed above is fundamentally a consequential question. These types of fundamental limitations in empirical approaches explain why they have not been adopted as regulatory tools.

4.3 The high ILUC-risk assessment as a form of attributional modelling

In the RED II, a new category of high ILUC-risk biofuel feedstocks was created (the context for this is discussed in section 6.1. This category is defined not by reference to economic modelling of ILUC emissions and the illustrative ILUC values included in Annex VIII of the Directive, but in relation to the historical link between expansion of each feedstock and conversion of high carbon stock landscapes to agricultural use. High ILUC-risk feedstocks are defined as feedstocks "for which significant expansion of the production into land with high-carbon stock is observed". These feedstocks are to be gradually excluded from eligibility to receive supply incentives. The high ILUC-risk framework seeks to use a result that can be established based on examination of historical data as a proxy for a more detailed consequential assessment of ILUC risk. Identifying a feedstock as high ILUC-risk involves making a claim something like this:

> *"Given that the feedstock being assessed has historically been associated with a high level of deforestation and/or peat loss, it is reasonable to assume that diverting some of this feedstock for biofuel use is likely to cause further feedstock expansion into those high carbon stock areas, and therefore a high level of indirect land use change emissions."*

One important difference in scope between the high ILUC-risk assessment and ILUC modelling is that the high ILUC-risk assessment excludes any consideration of carbon stock change in landscapes that could be characterised as having low or medium carbon stocks. The conversion of high carbon-stock areas is of course the most deleterious form of land use change, but conversion of shrubland or even grassland to agriculture can still result in carbon stock changes that are large compared to the GHG savings from fossil fuel displacement. Such more diffuse ILUC emissions are not considered in the high ILUC-risk assessment.

The high ILUC-risk assessment (European Commission, 2019) is based on a review of historical data on areas in which high carbon stock landscapes have been replaced by production systems for biofuel feedstock crops. Analysis by the European Commission (European Commission, 2019) referenced earlier studies documented in the academic literature and original satellite data analysis of deforestation and peat drainage in order to estimate the extent to which expansion of the main biofuel crops had resulted in conversion of high carbon stock land in the period 2008-17.

The high ILUC-risk assessment can be understood as an attributional assessment, in that it involves identifying an impact in the world (conversion of high carbon stock land), and then attributing that impact across a set of systems (additional area brought into production of a given crop). Unlike a conventional attributional LCA approach, however, the high ILUC-risk assessment does not readily allow all emissions incurred to be allocated to specific farms. This is because the change in area for a given crop is identified based on aggregate agricultural statistics rather than by identifying all of the specific areas where a crop is newly planted, as it is generally not possible to map the areas devoted to each individual crop in that level of detail (although the constant development of better remote sensing techniques and deployment of new satellites mean this will become more possible over time).

Having made an estimate of the fraction of expansion of each crop that occurs at the expense of high carbon stock land, the high ILUC risk assessment adjusts this fraction for the typical productivity of each crop before it is compared to a threshold value. This allows a fairer comparison of the potential land use change impact from one unit of production. If a cropping system is considered twice as productive as the baseline (normalised to the productivity of soybean cultivation) then the expansion fraction will be divided by two. These adjusted values are compared to the threshold which is set at 10%, and if the adjusted value is higher, then the feedstock is defined as high ILUC-risk. Currently, only palm oil is defined as high ILUC-risk, though the Commission has a mandate to update the assessment periodically.

5 Challenges and decisions in ILUC modelling

In section 3.1 we identified six basic factors in ILUC modelling (gross land demand; role of co-products; demand change; productivity; relocation; land use change emissions). A 'good' ILUC model can be thought of as one in which the balance of these factors is consistent with our best assessment of what is likely to happen in real agricultural markets. A 'bad' ILUC model, by contrast, would be one in which one or more of these factors was completely incompatible with our best assessment of what is likely to happen in real agricultural markets. If, for example, a model predicted the emergence of a large palm oil production industry in Denmark we would readily identify this as a bad model because we know that oil palms cannot grow in the Danish climate. Below, we briefly discuss some of the more important issues that ILUC models must address in order to achieve a credible balance between these factors, and then discuss some general challenges associated with ILUC modelling.

5.1 Parameterising agricultural markets

5.1.1 Market connections

Agricultural produce is traded globally, and the EU biofuel market is supplied with feedstocks produced in dozens of countries from a variety of crops. The links between agricultural and carbon stock changes are heterogeneous by geography and by crop, and therefore it matters whether a model assumes that the additional production to meet demand for biofuels occurs in the region with the biofuel mandate in the crop that is processed to biofuels, or whether it is globally distributed and the supply of a range of crops increases to meet demand.

As was discussed in section 3.4.3, the two main approaches to modelling the role of trade between countries are to assume a single world market in which production expands wherever it is most cost efficient, or to assume that there is a degree of inertia in bilateral trade patterns (Armington approach) and that countries will tend to import from the same countries that they have imported from in the past. There is no simple answer to which of these approaches is better, but as an example we can consider historical data on the development of vegetable oil and animal fat imports to the United States. Figure 14 shows the most significant six feedstock-country pairings by import quantity. Figure 15 then shows these same imports normalised against the level in 2004 (we chose 2004 for the example as a number of GTAP ILUC studies have been based on the 2004 GTAP database). Tallow imports from Canada are not shown in Figure 15 as they were zero in 2004 and therefore could not be normalised on that basis.

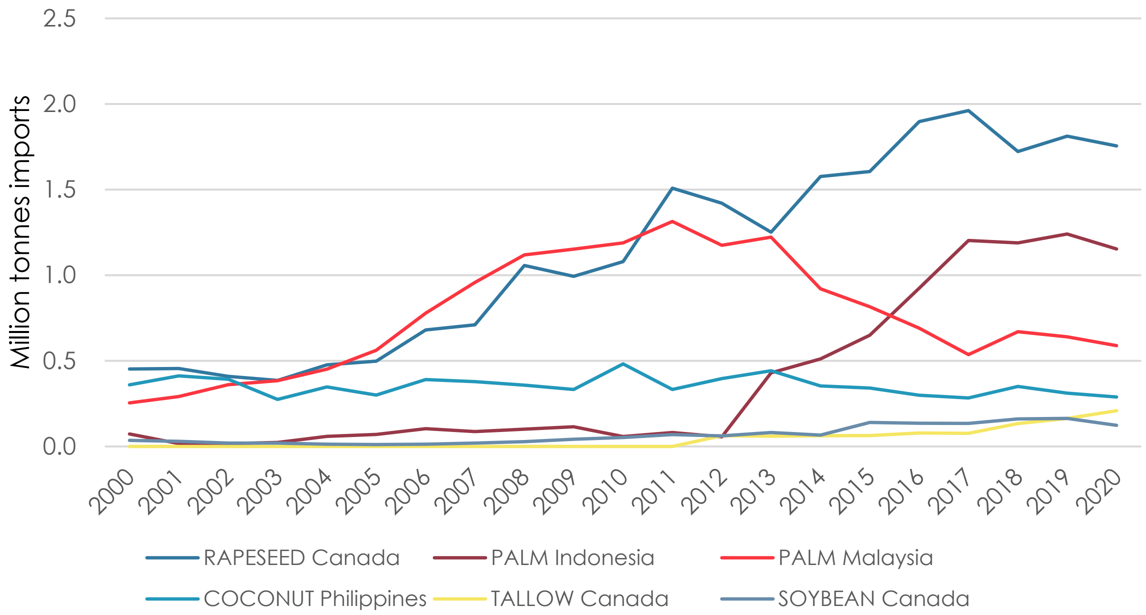

Figure 14 Major sources of U.S. vegetable oil/animal fat imports, 2000-2020

Source: US ITC (2021)

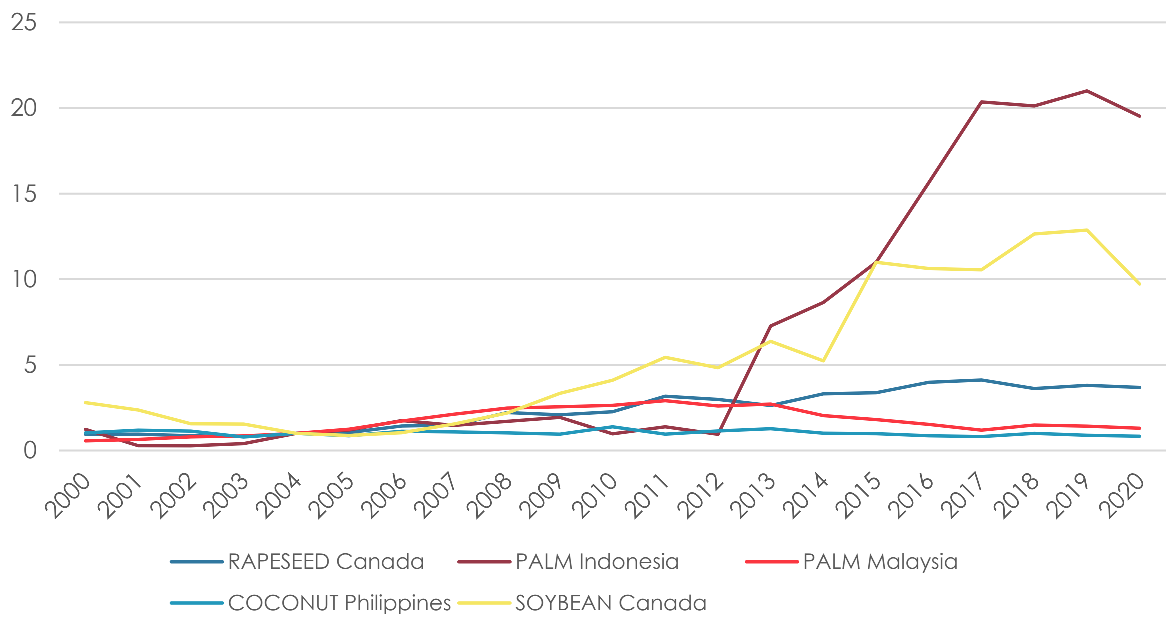

Figure 15 Imports of vegetable oils to U.S. normalised against 2004 level

Source: US ITC (2021)

The data show that in the period 2004 to 2020 the largest increases in vegetable oil imports were from Canadian rapeseed oil with nearly a factor four import increase, and Indonesian palm oil with a factor 20 import increase. This very rapid expansion of palm oil imports from Indonesia simply could not be predicted with an Armington approach, but clearly would be very relevant to ILUC modelling. Other combinations with an increase in imports by a factor of twenty or more (but lower absolute volumes) included Ukrainian, French and Dutch sunflower oil and Australian rapeseed oil. None of these fundamental changes in trade pattern could be predicted in an Armington framework. On the other hand, a single world market framework might have predicted trade from too many places – while new trade relationships have emerged, the overall volumes are still dominated by a small number of countries.

Related to changes in trade relationships is the question of how readily different similar products can replace each other in the market. Vegetable oils are all at least somewhat similar, but they have different properties that inform their relative pricing and market roles. A model that assumes that similar crops are readily fungible with each other can be expected to produce relatively similar ILUC results for similar feedstocks. A model that assumes that different vegetable oils or different grains cannot relatively substitute each other is more likely to produce big differences in ILUC outcomes. Again, these issues are difficult to precisely parameterise for a mathematical model.

5.1.2 Productivity responses

One of the central questions in ILUC modelling is the balance between 'intensive' and 'extensive' responses to increased commodity demand. Intensive responses include increased yield by greater use of inputs, increased yield by agronomic advancement, increased cropping intensity and switching to crops with higher yields. The extensive response is bringing new land into production. The hierarchy between these responses (whether most of the increase in supply comes from more land or from improved productivity) is crucial to ILUC assessment, but it is rather a hard question to analytically answer.

The great complicating factor in modelling productivity change is the difficulty of unpicking productivity improvements that are a response to demand and/or prices from productivity improvements that are a result of the background rate of progress and technological development. The yields for most crops have a remarkable tendency to increase in a linear fashion over time (Malins et al., 2014), and it has proved rather difficult to convincingly demonstrate whether or not this is driven by increased demand. On the one hand, high prices make more resources available to invest in productivity improvement. On the other hand, low prices could focus farmers and governments on finding ways to improve output. Econometric analysis can be used to investigate whether historical yields have shown a response to price, but there is disagreement in the field about what the historical data really show (cf. Berry, 2011). Undertaking robust historical analysis is made difficult because of the limited number of datapoints when there is only one crop per year, and because it is essentially impossible analytically to unpick short term (one season) responses from longer term responses – how does one work out the relative contributions to yield recorded in 2020 from: innovations that were researched in response to high prices in 2007; farm equipment that was bought when prices were high in 2011; extra fertilisers applied because futures prices were high when the crop was planted; and technological improvements that had nothing to do with whether prices were high or low? On the area side of the question. while there is clear evidence that area for a given crop is more responsive than yield to prices in the short term, expansion of one crop is not the same as expansion of agricultural area overall – the area of rapeseed farmed in Europe might increase to produce more biofuel feedstock, but it's difficult to identify what impact this has on total agricultural area when other crops are replaced, especially if the overall trend is for reduction in total area (so that the impact may be to reduce farmland abandonment rather than increase farmland expansion).

5.2 Decisions in the face of inadequate data

In the ideal world, the parameters in ILUC models would be informed by a combination of detailed analysis of historical data to identify relationships that have existed in the past and detailed agricultural modelling to identify what will be possible in the future. In practice, however, the availability of robust analysis is very limited for some of the most important questions modellers face. Take the parameterisation of yield increase in the GTAP general equilibrium model. In 2009 when the model was used for the first regulatory ILUC assessment for the California Air Resources Board, the response of crop yields to price changes for all crops of the world in all regions of the world was set based on values estimated for the U.S. corn crop. Not only was there little or no direct analysis to draw on to confirm that U.S. corn was representative of other crops in the U.S., never mind other crops in other regions, but even the analysis used to set the U.S. corn value was robustly critiqued (Berry, 2011). The reality of equilibrium modelling is that while modelling choices may be informed by the modellers' expert understanding and by such data as is available, in the end they are just that – choices. Assessing which data to treat seriously, deciding when analysis of one region can reasonably been used as a proxy for others, deciding which crops should be in the same nest as each other in the general equilibrium models and which production options are important enough to include in the partial equilibrium models are all decisions that are data informed but, in the end, subjective. Modellers end up undertaking iterative processes behind the scenes tweaking the balance of parameters to produce outcomes that they consider realistic, or just to make sure that the models are able to come to analytical solutions at all. The role of expert judgement in ILUC modelling is unavoidable, but it means that outcomes can be more sensitive to the expectations of the modellers than one might like in a truly objective process. Comparing ILUC frameworks is therefore more than just a matter of considering the analytical underpinnings and individual model inputs.

6 Dealing with ILUC through renewable fuel policy

The various ILUC modelling approaches discussed above have generated a wealth of evidence that suggests that ILUC is a real and significant problem – but having identified ILUC as a concern, the question inevitably arises of how regulation might be used to reduce or avoid ILUC emissions. Addressing ILUC in policy is made particularly challenging by the lack of certainty that is associated with ILUC modelling, and the lack of consensus in the stakeholder community. Stakeholders on all sides of the argument are able to point to research and expert opinions to support the range of sometimes mutually contradictory positions they espouse. While anything resembling general agreement on the ILUC problem can be expected to remain elusive for the foreseeable future, a general recognition has emerged over the past decade that ILUC cannot simply be ignored, and that some sort of policy response must be considered.

6.1 An abridged history of the ILUC discussion in the EU

When the first RED was approved in 2008, it was explicitly recognised in the text that ILUC was a problem that needed to be addressed. The 85th recital states that,

> *"The Commission should develop a concrete methodology to minimise greenhouse gas emissions caused by indirect land-use changes. To this end, the Commission should analyse, on the basis of best available scientific evidence, in particular, the inclusion of a factor for indirect land-use changes in the calculation of greenhouse gas emissions and the need to incentivise sustainable biofuels which minimise the impacts of land-use change and improve biofuel sustainability with respect to indirect land-use change."*

The RED required the Commission to produce a report "reviewing the impact of indirect land-use change on greenhouse gas emissions and addressing ways to minimise that impact," which was to be accompanied (if appropriate) by a proposal for legislative amendments to the RED. To inform this report the European Commission generated four studies. One was led by the Italian branch of the Joint Research Centre (Edwards et al., 2010) and presented a comparison of ILUC modelling results from several economic models, both partial and general equilibrium. A second study led by the Spanish branch of the Joint Research Centre (Blanco Fonseca et al., 2010) presented a set of partial equilibrium modelling results. A third undertaken in-house by DG Energy provided an extensive review of existing literature on indirect land use change and associated agronomic issues (European Commission Directorate General for Energy, 2010). Finally, DG Trade asked the International Food Policy Research Centre (IFPRI) to undertake general equilibrium modelling with MIRAGE (Al-Riffai et al., 2010).

The release of these four reports was followed by a protracted period of debate both within and beyond the Commission[25]. A stakeholder consultation held in 2010[26] received over a hundred responses from a broad range of organisations, and the JRC held an expert workshop in November 2010 (Marelli, Mulligan, et al., 2011). The discussion was characterised by fundamental disagreements and differences of perspective between different stakeholder groups. This included extensive discussion over the validity of various elements of the ILUC modelling frameworks, and for some stakeholders a persistent refusal to acknowledge that ILUC impacts (or indeed food price impacts) were a real concern.

At the end of 2010, the Commission released the required summary report (European Commission, 2010). At that time, the Commission concluded that:

- "It can be argued that the best available methodology to estimate (indirect) land-use change is still through economic models where decisions are made based on relative prices."
- "A number of deficiencies and uncertainties associated with the modelling, which is required to estimate the impacts, remain to be addressed, which could significantly impact on the results of the analytical work carried out to date."
- "Indirect land-use change can have an impact on greenhouse gas emissions savings associated with biofuels, which could reduce their contribution to the policy goals, under certain circumstances in the absence of intervention."

After nearly another two years (and following an update by Laborde (2011) to the MIRAGE ILUC analysis) the Commission released its proposal for what was to become the 'ILUC Directive' (European Commission, 2012). The process of agreeing this proposal within the Commission was itself contentious, with significant differences in attitude emerging between different directorates general. The final proposal included the following changes to the RED framework:

1. Limiting the contribution of food-based biofuels to the 10% target for renewable energy in transport in 2020 to 5 percentage points.
2. Introducing a category of advanced biofuels to be quadruple counted towards compliance with RED targets. This would apply to the feedstocks in what has become part A of Annex IX of the RED, with the exception of generic categories for cellulosic and ligno-cellulosic material which are now on part A of Annex IX but were proposed for inclusion in part B, which was to be only double counted.
3. Strengthening the requirement on the minimum GHG savings for biofuels calculated under the RED LCA methodology.
4. The introduction of ILUC factors to be used in reporting to Member States by suppliers and by Member States to the European Commission, but not in relation to the mandatory minimum GHG saving requirements for biofuels or to the assessment of GHG intensity reductions required in the Fuel Quality Directive.

[25] Indeed, the disagreements within the Commission around the first RED were adequately noteworthy that there is an entire academic paper focused on the role of one policy official in driving the process forward (Sharman & Holmes, 2010).

[26] There was also an early 'pre-consultation' held in June 2009.

An earlier leaked draft of the proposal included a more interventionist policy platform than was eventually adopted, including Malins & Searle (2012):

5. The integration of ILUC factors into the FQD LCA methodology.

6. A commitment to phase out support for food-based biofuels after 2020.

Let us briefly review the intentions and potential impacts of these six measures.

The first measure, the introduction of a cap on support for food- and feed-based[27] biofuels (finally agreed at 7% in the adopted ILUC Directive), was intended to limit both the ILUC impacts and food market impacts of EU biofuels use. The press release for the proposal included a quote attributed to the Commissioner for Climate Action at the time stating that, "We must invest in biofuels that achieve real emission cuts and do not compete with food. We are of course not closing down first-generation biofuels, but we are sending a clear signal that future increases in biofuels must come from advanced biofuels. Everything else will be unsustainable." The food cap is a relatively blunt instrument although as discussed below (in section 6.3) the RED II creates the possibility for Member States to differentiate treatment of food-based fuels within the cap.

The second measure, quadruple counting for listed feedstocks and for renewable fuels of non-biological origin (electrofuels), was intended to, "encourage a greater market penetration of advanced (low-ILUC) biofuels". This multiple counting could be thought of as a 'carrot' for advanced biofuel production to go with the 'stick' of the food cap. Quadruple counting could in principle have been a strong value incentive for development of advanced fuel technologies. Compliance credits under national RED schemes tended to have a value around 20 to 30 €cent per litre of fuel supplied, on which basis a quadruple credit would be worth about 1 € per litre. The intention was that driving development of the advanced biofuel market would have allowed these to supersede food-based first-generation fuels. However, the quadruple counting measure did not achieve wide support in the stakeholder community for a couple of reasons. Firstly, multiple counting of credit multiplies not only the value available but the value uncertainty. If the basic value of RED compliance is 30 €cent per litre the quadruple credit on one litre of fuel is worth €1.20, but if the basic credit falls to 5 €cent per litre the value of the quadruple credit plummets to 20 €cent. Advanced biofuel producers want value confidence in order to make investments, and quadruple credits did not seem to offer this. Secondly, quadruple counting introduces considerable uncertainty into the total amount of renewable fuel needed to meet targets, and in principle could allow advanced fuels to aggressively displace first generation fuels – an extra million litres of advanced fuel would reduce by 4 million litres the requirement under the RED to supply first generation material. This would exaggerate the inconsistency between the amounts of fuel needed to comply with 2020 targets under RED and FQD (where quadruple counting would not have applied) and make it harder for civil servants to predict the overall contribution of biofuels to meeting national CO_2 inventory targets. This potential enhanced competition between advanced and first-generation biofuels was also predictably unpopular with existing biofuel companies, including companies with interests in both advanced and first-generation technologies. In the adopted ILUC Directive, quadruple counting was retired

[27] For brevity, in the rest of this report we abbreviate 'food- and-feed based' to 'food based'. It should be understood however that the grain crops (primarily wheat and corn) used for biofuel production are generally 'feed grade' and thus it can be reasonably argued that the direct competition for these is primarily between biofuels and livestock feed rather than biofuels and food for direct human consumption.

in favour of the creation of an indicative target for advanced biofuel supply. It is impossible to know now how effective the quadruple counting mechanism could have been, but the indicative target has done little to move advanced biofuel technologies to commercial scale.

The third element of the proposed policy was to increase the minimum required reportable GHG savings. The RED defines the GHG saving from a fuel as the percentage difference between its attributionally calculated GHG emission intensity and the GHG intensity set for a fossil fuel comparator. We describe this as a 'reportable' GHG saving in order to emphasise that the difference between these two attributionally calculated GHG emissions intensities may not be a good approximation for the actual net GHG impact of increasing the use of the biofuel in question if it were calculated on a consequential basis.

The intention of increasing the minimum GHG saving requirement was to reduce the expected direct emissions from the biofuel supply chain and thereby improve the emissions performance of the policy as a whole. Implicit in this is the idea that if ILUC emissions will offset and potentially eliminate the savings from fossil fuel displacement, then a tougher threshold will reduce the likelihood that the policy overall does more harm than good. This logic can also apply to other areas of uncertainty and variability in the emissions assessment, such as nitrous oxide emissions from nitrogen fertilisation which can also be quite significant.

The fourth element of the proposal, reporting of ILUC factors but without effect on compliance with sustainability requirements or contribution to the FQD target, can be considered more symbolic than directly impactful, and reflects a compromise between those who felt that ILUC factors should be made fully part of the lifecycle assessment and those who felt they should be excluded from the Directive entirely. The ILUC factors listed are split into three feedstock categories (cereals and other starch rich crops, sugars, and oil crops), calculated based on the feedstock-specific values in Laborde (2011). The reporting of ILUC emissions does not directly change the value proposition for biofuel producers. Some fuel suppliers, especially those with more reputational exposure, may be sensitive to the optics supplying biofuels that deliver no net emission reduction if the ILUC factor is taken into account. It is possible that this has contributed to increased sourcing of waste feedstocks for biodiesel, but we are not aware of any firm evidence demonstrating a change in behaviour due to this measure. The reporting of ILUC factors may have more impact through its effect on Member State policy making – it ensures that ILUC emissions receive characterisation in policy studies and impact assessments.

The fifth measure (which was included in the draft proposal but removed from the final version) would have made ILUC factors part of the regulatory LCA in the FQD, and therefore affected the contribution of food-based biofuels to the 6% transport GHG intensity reduction target for 2020. This would have made the lifecycle accounting under the FQD similar to that in the California Low Carbon Fuel Standard (see section 6.2.2). Including ILUC accounting in FQD would have meant that several country's fuel suppliers needed to fundamentally revise the biofuel feedstock mixes to meet the requirement. Figure 16 shows that based on their 2019 feedstock mixes inclusion of ILUC accounting would have reduced reportable GHG reductions by at least half for Austria, Belgium, Czechia, Denmark, Latvia, Lithuania, Luxembourg, Poland, Slovakia and Spain.

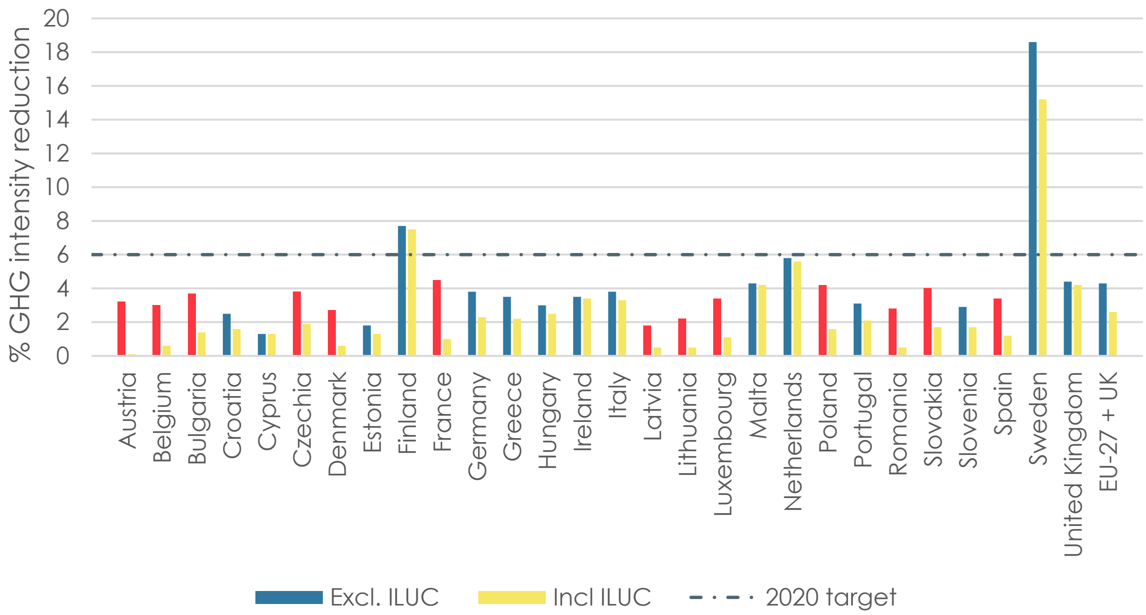

Figure 16 **GHG intensity performance by country in 2019 as compared to the 2020 target for a 6% reduction, with and without ILUC accounting. Red bars indicate countries where the GHG savings reported with ILUC are less than half those reported without.**

The sixth measure, also removed from the final proposal, was a commitment to eliminate support for food-based biofuels after 2020. As proposed, this would have been a strong signal for the shift to advanced biofuels, although given how firmly this would have been resisted by many Member States and stakeholders it may not have been treated as a solid basis to invest even had it been passed (i.e. if this language had been in the final agreed ILUC Directive, investors in advanced biofuels would have been concerned that the commitment would be softened again later).

The final important development in the EU's ILUC knowledge base following the adoption of the ILUC Directive but prior to the adoption of the RED II was the commissioning of additional ILUC modelling, this time using the GLOBIOM partial equilibrium framework (Valin et al., 2015). In particular, this work supported the basic conclusion from the previous MIRAGE work that vegetable oils are likely to have higher ILUC emissions than ethanol feedstocks. It also added results on potential ILUC emissions from the cultivation of cellulosic biomass crops and short rotation coppice, which were not considered by Laborde (2011).

6.1.2 The ILUC factors in Annex VIII of the RED I and II

The ILUC Directive led to the inclusion of Annex VIII of the RED I of "provisional estimated indirect land-use change emissions" by feedstock group. These are shown in Table 1. The ILUC values given here are based directly on the feedstock specific results in Laborde (2011) from general equilibrium consequential modelling with MIRAGE. A weighted average is taken for each feedstock group (wheat and corn in the cereals group, sugar beet and sugar cane in sugars and palm, soy, rapeseed and sunflower oil in the oil crops).

Table 1 Provisional estimated indirect land-use change emissions in the RED I and II

Feedstock group	Mean (²)	Interpercentile range derived from the sensitivity analysis (³)
Cereals and other starch-rich crops	12	8 to 16
Sugars	13	4 to 17
Oil crops	55	33 to 66

These values remain the same in the RED II, despite the publication in the interim of GLOBIOM modelling in Valin et al. (2015) and the use of those more recent values in parts of the RED II impact assessment.

6.1.3 Current status – ILUC in RED II

The approved ILUC Directive (European Union, 2015) set a direction for EU policy on ILUC that is continued with the RED II. Firstly, the RED II maintains a cap on the use of food-based fuels. This is set at whichever is lower between 7% of transport energy[28] and 1% of transport energy plus the amount supplied in 2020, but may be further reduced by the individual Member States. Secondly, it significantly strengthens support for advanced biofuels in Europe by including a binding sub-target for the use of biofuels from feedstock on Part A of Annex IX. The European Commission's position remains that growth in biofuel consumption must be delivered through more advanced technologies and low value feedstock resources.

Land use change accounting is now addressed through three regulatory mechanisms in the RED II. Firstly, the lifecycle analysis requirements used for assessing compliance with minimum GHG savings thresholds retain the attributional approach of the RED I, requiring that land use change emissions must be accounted if the use of the land associated with a given feedstock batch has changed since January 2008. Secondly, Annex VIII of the Directive still includes "provisional estimated indirect land-use change emissions from biofuel, bioliquid and biomass fuel feedstock" as listed in Table 1. These values can be added to the reported attributional emissions to create a hybrid LCA metric and allow ILUC to be reflected when undertaking policy analysis, but do not affect compliance with the required minimum reportable GHG savings. Thirdly, a category of high ILUC-risk biofuel feedstocks has been created, defined as feedstocks "for which significant expansion of the production into land with high-carbon stock is observed", identified using the approach outlined in section 4.3. Currently palm oil is classified as high ILUC-risk and will no longer be eligible for support by 2030 (and some Member States are removing support earlier). As we will discuss in more detail in section 6.3, RED II also adds explicit leeway for Member States to impose additional differentiation between biofuels based on the best available evidence on ILUC emissions.

[28] In the RED and RED II only transport energy for road and rail are considered when assessing these caps and energy supply targets.

6.2 Examples of ILUC regulation from the United States

Before we return to the scope for Denmark to address ILUC with additional measures in the national implementation of RED II, it is useful to briefly review the handling of ILUC in North America. While European legislators have always baulked at the prospect of fully including ILUC emissions in the regulatory lifecycle analysis requirements of the RED and FQD, this has not been the case in the United States where ILUC emissions have been integrated fully as part of LCA requirements for over a decade. This is true at both the federal level under the national Renewable Fuel Standard (this is the second federal renewable fuel standard, abbreviated as 'RFS2'), and at state level for California through its Low Carbon Fuel Standard (LCFS) (and more recently in Oregon which has ILUC emissions included in the LCA under its Clean Fuel Program, which is based on the LCFS regulatory structure).

6.2.1 The U.S. Renewable Fuel Standard

The RFS2 sets annual volume targets for the supply of biofuels in the U.S., with tiers of support intended to provide a stronger value proposition to cellulosic and 'advanced' fuels that are expected to deliver better GHG performance than corn ethanol. The legal basis for the RFS2 was set out in 2007 by the Energy Independence and Security Act (EISA, U.S. Congress, 2007). Unlike the RED which does not include ILUC emissions in its LCA requirement, the EISA states that (our emphasis), "The term 'lifecycle greenhouse gas emissions' means the aggregate quantity of greenhouse gas emissions (***including direct emissions and significant indirect emissions such as significant emissions from land use changes***), as determined by the Administrator." The Environmental Protection Agency (EPA) as administrator determined that ILUC emissions must therefore be included in the assessment. This was done (as discussed in section 3.2.2) with the FAPRI-FASOM partial equilibrium modelling framework. Unlike California and Europe, rather than adopting a hybrid approach to agricultural emissions by using an attributional assessment for emissions from farming and a consequential assessment for ILUC, the EPA approach also uses FAPRI-FASOM to determine emissions from farming, including fertiliser use, change in methane emissions from rice paddies and change in methane emissions from livestock.

Support for biofuels under the RFS2 is conditional on meeting certain GHG emissions reduction thresholds. To be classed as a 'renewable fuel' (the lowest tier of support) a biofuel must be assessed as delivering 20% emissions reduction or better compared to a petroleum comparator value or be produced at a facility already operational prior to the adoption of the EISA[29]. To class as an 'advanced' fuel or a biomass-based diesel fuel (categories with a greater associated value of support) a biofuel must be assessed as delivering at least 50% emission reduction compared to petroleum fuels. ILUC emissions are included in this comparison. The EPA determined that corn ethanol from typical (natural gas fired) facilities delivered slightly better than a 20% GHG emission reduction and thus most corn ethanol facilities qualify to contribute to the renewable fuel targets[30]. Soy biodiesel delivers slightly better than a 50% GHG emissions reduction

[29] This is a 'grandfathering' clause intended to protect existing less efficient biofuel facilities (primarily corn ethanol plants, although the grandfathering clause has also been used to allow some palm oil biodiesel to be counted as renewable fuel under RFS2).

[30] Corn ethanol is considered ineligible under RFS2 to count as an advanced fuel even if it was able to meet the 50% GHG saving standard.

and therefore qualifies as an advanced biomass-based diesel. For these main compliance fuels (i.e. the fuels supplied in the largest volumes in the U.S) the inclusion of ILUC emissions therefore did not affect the support available.

The one fuel pathway for which the inclusion of ILUC has been determinative of compliance status is palm oil biodiesel. In 2011 the EPA proposed a pathway determination which showed palm oil achieving less than the mandatory minimum 20% GHG saving (U.S. EPA, 2011), therefore excluding palm-oil-based fuels from counting towards any of the RFS2 targets unless counted as a renewable fuel through grandfathering. The inclusion of ILUC emissions has therefore strongly curtailed the opportunity for palm-based biofuels in the U.S.

6.2.2 The California Low Carbon Fuel Standard

The California LCFS was introduced at the same time as RFS2, adding an additional layer to alternative fuel regulation in California (fuels supplied in California are still eligible to also be counted towards RFS targets). The LCFS is like Article 7a of the Fuel Quality Directive in that it sets GHG intensity reduction targets for transport fuels supplied in California, but unlike the FQD (which relies on Member States to set implementation details and interim trajectories) the LCFS has annual targets, a clear penalty structure and a system of credit award and trading. GHG emission reductions under the LCFS are calculated on a hybrid LCA basis. An attributional LCA score is determined based on the CA-GREET lifecycle analysis tool (CARB, 2018) and a consequential ILUC score is added to it based on ILUC modelling undertaken by CARB using the GTAP model. This GHG emission intensity is then compared to the assessed average GHG intensity of fossil fuels supplied in the state (with separate values for petrol and diesel substitute fuels); for each unit of fuel supplied, credits are awarded in proportion to the calculated value for the GHG saving.

Because the number of credits awarded under the LCFS is directly affected by the ILUC emissions for each pathway, the ILUC scores have a more direct bearing on the value proposition than they do under the RFS2 (where all fuel pathways determined to date, except palm oil biodiesel, have met the threshold for the expected tiers of support). Also unlike the RFS2, the ILUC scores under the LCFS have been revised once since the original calculations – this resulted in a reduction of reportable ILUC emissions for all pathways considered. LCFS does share with the RFS2 the calculation of a relatively high ILUC value for palm oil – the LCFS's assessed ILUC emissions of 71 gCO_2e/MJ[31] for palm oil biodiesel are so high as to make it very difficult for palm-oil-based fuels to deliver any significant credit generation under the LCFS. Fuels that are given an ILUC value of 0 gCO_2e/MJ (such as fuels from wastes and residues and cellulosic fuels) therefore have a value advantage under the LCFS.

[31] This value is quoted on the California time accounting convention of 30-year amortisation – it therefore differs from the value shown in Figure 10 which has been adjusted to a 20-year basis which is the standard EU time accounting convention.

6.3 Regulating ILUC under the Renewable Energy Directive II: Article 26(1)

As discussed above, the RED II sets several clear requirements for Member States relating to the way that biofuels may be supported and counted towards national targets, including the food cap, ILUC factor reporting and the sub-target for advanced fuels. These are not, however, the only measures that Member States are permitted to consider in order to minimise ILUC emissions. Paragraph 1 of Article 26 of the Directive states that (our emphasis):

> *The share of biofuels and bioliquids, as well as of biomass fuels consumed in transport, where produced from food and feed crops, shall be no more than one percentage point higher than the share of such fuels in the final consumption of energy in the road and rail transport sectors in 2020 in that Member State, with a maximum of 7 % …*
>
> *Member States may set a lower limit and* ***may distinguish, for the purposes of Article 29(1), between different biofuels****, bioliquids and biomass fuels produced from food and feed crops,* ***taking into account best available evidence on indirect land-use change impact****. Member States may, for example, set a lower limit for the share of biofuels, bioliquids and biomass fuels produced from oil crops.*

The text allows Member States to distinguish between fuels on the basis of the best available evidence on ILUC emissions, but it is important to note that this basis is provided in the context of setting a cap on the contribution of food-based fuels more generally. The example given makes it clear that Member States are entitled to set a lower cap on some food-based biofuels than others, but it is less clear which other ways of distinguishing between fuels may or may not be allowable under this paragraph. It should be noted that because Article 26(1) refers specifically to food- and feed-based biofuels we believe that this text does not provide a basis to distinguish between non-food feedstocks on the basis of expected indirect emissions. This would preclude, for example, distinguishing between biofuels from waste or residual materials on the basis of expected indirect emissions due to displacement from other uses (cf. Malins, 2017b) or distinguishing between cellulosic biomass crops based on modelled ILUC emissions (cf. Valin et al., 2015).

The text says that biofuels may be distinguished "for the purposes of Article 29(1)". Article 29(1) states that biofuels may only be eligible for financial support and counted towards EU targets if they meet the defined sustainability criteria of the Directive. Article 29(1) does not mention the food cap, which suggests that distinguishing for the purposes of Article 29(1) could go beyond setting tighter caps. Indeed, we understand from the interaction of these articles that Article 26(1) provides a broad basis for Member States to differentiate the financial support available to biofuels depending on best available evidence regarding ILUC emissions.

The second question regarding Article 26(1) is what should be understood by the 'best available evidence on ILUC impact'. This is explored further in the next section.

7 Identifying the best evidence on ILUC emissions

The use of the term best available evidence echoes the clause in Article 19(6) of the RED I that required the Commission to review best available scientific evidence on ILUC. It also echoes the requirements set on the Commission in RED II in relation to the delegated act identifying high ILUC-risk biofuels, which was to be based on 'best available scientific data'. As was referred to in section 6.1, the last time the European Commission directly adjudicated on what constituted best available scientific evidence on indirect land use change was in its report on indirect land use change for RED I and subsequently in the explanatory memorandum on the proposal for an ILUC Directive. The report (European Commission, 2010) states that "it can be argued" that the best available evidence is that from economic ILUC models. This is confirmed in the proposal for the ILUC Directive (European Commission, 2012) which states that the estimated ILUC values included in Annex VIII (derived from the MIRAGE modelling) are "based on the best available scientific evidence". Further review of the best available evidence on ILUC is provide in the 2017 review of ILUC studies for the Commission (Woltjer et al., 2017).

Based on these documents, it can be concluded that the results of economic ILUC modelling and of the high ILUC-risk assessment are considered by the European Commission to be part of the body of best available evidence. While the ILUC values in Annex VIII of RED II are based on Laborde (2011), there is nothing to prevent other evidence from being considered. It is relevant that the impact assessment for the RED II (European Commission, 2016) included the use of results from the GLOBIOM model (Valin et al., 2015), from which it may be concluded that the Commission considers these results to be comparably credible to the MIRAGE results.

Throughout the ILUC debate, the Commission has relied preferentially on results produced by the studies it has directly contracted, and has not given equivalent weight to results of studies undertaken for other institutions or focusing on biofuel mandates in other parts of the world. This includes discounting analyses that have been undertaken by biofuel producers in the hope of arguing that their likely ILUC emissions are lower than the results from the main Commission modelling.

While there is no explicit barrier to a Member State considering additional evidence in order to develop measures under Article 26(1), it can be reasonably assumed that if the Member State came to conclusions that were strongly inconsistent with those from the Commission's MIRAGE and GLOBIOM modelling, and the high ILUC-risk assessment, that this may be considered problematic by the Commission and would need to be very well justified. It would clearly be counter-productive to have a patchwork of ILUC treatments across the EU with different feedstocks favoured in different Member States, both because it would impose a burden on fuel suppliers and because it could encourage a shuffling whereby the supply of biofuels to each Member State could be optimised to meet local rules without delivering any meaningful change in the overall feedstock mix for the EU, and therefore without any significant reduction in ILUC emissions.

7.1 Development of new ILUC values for use in determining levels of support

An exemplar for the regulation of ILUC is provided by the State of California. California has contracted ILUC modelling to produce ILUC factors for a number of potential biofuel feedstocks and has integrated these values into the lifecycle analysis of fuel supplied under the LCFS through a hybrid LCA approach. Denmark could consider following a similar route by developing new ILUC modelling and using the results of that modelling as a basis for providing support to biofuels.

Given that there is already an extensive literature of ILUC results available, what advantage might be delivered by undertaking new modelling? Firstly, both land use change and the study of land use change are dynamic. New modelling could potentially take advantage of developments in understanding that have been achieved over the past six years. The most important and difficult questions in ILUC modelling remain much the same as they have been for the past decade – questions about the relatively roles of intensive and extensive responses in meeting additional agricultural demand – and these have not been finally resolved, but a new exercise could acknowledge and integrate more recent research.

Secondly, a Danish ILUC modelling programme could provide an additional opportunity for Danish stakeholders to be engaged through consultation. If handled well, an active open consultation could help build support in the Danish community for additional regulatory action on ILUC and would be an opportunity to consider any particularities relating to the Danish situation.

Thirdly, a new ILUC modelling exercise would be an opportunity to choose the set of research questions to be considered, and potentially get answers that are either absent or obscure in the previous work. This could include producing more detailed outputs detailing predicted price changes, more detailed decompositions to help explain results, modelling additional feedstocks, or further differentiating between ILUC associated with feedstock from different markets.

Adding further disaggregation to model results may be of interest because In the ILUC analyses undertaken for the European Commission to date single ILUC values have been reported for each feedstock without attempting to distinguish between country of origin. This means, for example, that the palm oil ILUC value reflects the results of increasing palm oil demand from the EU to the world market as a whole. The decision about where palm oil will be sourced is made endogenously in the models based on existing trade patterns. It would be possible to adjust the shock structure in the modelling with a view to distinguishing between the impact of sourcing palm oil from different countries. In practice, this would mean shocking the model by both increasing palm biodiesel consumption in the EU and forcing an equivalent increase in palm oil imports to the EU from each potential source market in turn. While such results would be interesting, producing meaningful outcomes at this level of disaggregation could be challenging. ILUC models can be expected to work best when considering marginal changes to existing trade patterns. If instead we consider the impact of creating a new and significant source of demand for a commodity from a country that is not currently a major exporter it may stretch the capacity of the framework – for instance we discussed in section 3.4.3 that the Armington model of international trade is not able to deal well with fundamental shifts in trade flows. At the minimum, adding additional disaggregation to modelling tools in this way would require a significant investment into additional model testing and development.

Similarly, if using a framework in which the EU is characterised in enough detail to distinguish individual Member States, in principle one could produce ILUC values for individual Member States. In practice, however, many of the modelling frameworks which could be used may lack the detail in their representation of the EU to distinguish between Member States, and given the free movement of goods in the internal market one would not expect large differences between a shock introduced in Denmark and one introduced in another EU nation. Adding further disaggregation of either the source or destination countries might not be the most effective use of resources, compared to investing in improving core modelling choices.

7.1.1 Would the development of new model results be worthwhile?

While undertaking new modelling work has some appeal, it also presents some very significant challenges. As a practical question, one would have to consider which modelling teams and modelling tools might be able and willing to take such work on. There would be an obvious advantage to working with the MIRAGE or GLOBIOM teams given that those models have already benefitted from extensive development to handle biofuel market sin the European context. This would mean working with the International Food Policy Research Institute (IFPRI, for MIRAGE) or with the International Institute for Applied Systems Analysis (IIASA, for GLOBIOM). While working with these institutions would be appealing, it is not at all guaranteed that either institution would have the capacity and/or interest to take on ILUC work for the Danish government. While both institutions employ teams of economic modellers, only a handful have worked directly on the European ILUC analysis, and the ILUC modelling by each was been led to a great extent by a single individual. To the best of our knowledge IFPRI has not undertaken any new ILUC modelling with MIRAGE in some years, and a cursory review of his Google Scholar listing shows that David Laborde (the previous lead modeller) has been working on a range of topics outside of bioenergy in recent years. Even if funding for new work was made available, IFPRI might not see it as a priority to take such work on. On face value the prospects of working with IIASA and GLOBIOM would be better as the GLOBIOM framework is still being actively used and developed in the context of developing ILUC estimates for CORSIA. The flipside of this is that it already represents a considerable commitment of modelling resources by IIASA, and they may not be likely to accept an invitation to develop variant modelling for Denmark, even with an offer of proportionate funding.

If it was not possible to agree a work programme with IFPRI or IIASA, Denmark would have to consider inviting bids for work from other researchers. Other tools are available that could potentially be used to assess question in the EU context, such as the general equilibrium models MAGNET (Modular Applied GeNeral Equilibrium Tool, cf. Philippidis et al., 2018), GTAP (see section 3.3.3), or DART (Dynamic Applied Regional Trade Model, Kiel Institute for the World Economy, 2020), or the partial equilibrium models Aglink or CAPRI (Blanco Fonseca et al., 2010). None of these models has been subject to the same type of programme of consultative development and stakeholder engagement in the European context that has been seen for IFPRI and MIRAGE, and the associated modelling teams may also be limited by the conflicting priorities that we suspect might affect the availability of the teams at IFPRI and IIASA. While it may be possible to agree a contract to develop new ILUC values for Denmark with one of these models for moderate cost and commitment (for example an investment on the order of €100,000 and on a timescale of six months to a year), without a more long-term commitment and additional expenditure it would be difficult to deliver results that could credibly be identified as improvements over the existing studies.

Aside from the practical challenges of getting high quality work agreed and undertaken, new modelling presents other potential drawbacks, especially if undertaken in an entirely new modelling framework. For one, A new modelling exercise would be expected to deliver results with at least some differences at the feedstock level from the previous studies. Some divergence in results is fine from an academic analytical viewpoint – indeed, given the many sources of variation in ILUC modelling it would be highly surprising if a new modelling framework came very close to recreating outcomes of a given previous study. From the point of view of developing regulatory approaches, however, adding another set of ILUC values may do little to bring clarity to the market, especially if the modelling framework used was perceived to be limited compared to MIRAGE or GLOBIOM.

Beyond the challenges of procuring high quality work, presenting new results and proposing to use those new results as a basis for ILUC regulation would be likely to be a source of conflict with the European Commission. Commission officials are likely to be very reluctant to acknowledge new work undertaken outside of their control as constituting the best available evidence on ILUC emissions. This would represent both practical and political considerations. Practically, it may be difficult for Commission officials to invest the time required to fully evaluate new work. Doing so in relation to results from a single Member State would be difficult, doing so across 27 Member States could be practically impossible without significant reallocation of staff resources. Politically, even if a particular piece of new work was seen to be of high quality, the Commission may be reluctant to endorse work done by any single Member State. Firstly, it might be seen as effectively opening the door to lower quality work elsewhere. Secondly, the Commission may be cautious of supporting the creation of an analytical counterweight to its own work as it might be perceived as ceding an element of control of the EU narrative on ILUC to third parties.

Overall, while a new, well resourced, medium to long term programme of ILUC research would have great appeal as a basis to build on the ILUC evidence base, a very significant resource commitment would be needed to deliver clear added value on existing work. It is difficult to see that a new single study undertaken for Denmark, even if producing outputs of comparable quality to previous work for the Commission, would resolve any of the challenges with which policy makers are currently faced.

7.2 Based on ILUC values from the literature

If not commissioning new ILUC modelling work, policy could be based on the best available evidence drawn from ILUC results in the existing literature. As indicated above, we consider four sources to be the most relevant in this regard: the MIRAGE modelling in Laborde (2011); the GLOBIOM modelling in Valin et al. (2015); the ILUC review by Woltjer et al. (2017); and the high ILUC-risk assessment (European Commission, 2019).

7.2.1 Rely on ILUC values from MIRAGE work (Laborde, 2011)

The ILUC values from the MIRAGE work form the basis for the values by feedstock group given in Annex VIII of the RED II, and therefore it would be very defensible for a Member

State to treat either those group-level averages or the underlying feedstock level values they are based on as the basis for distinguishing fuels by ILUC impact.

The counter argument for focusing solely on these values is that it is now a decade since the work was undertaken, and therefore referring only to the MIRAGE work could imply discounting a considerable number of more recent studies. One answer to this is that work in the period since has tended to support the basic hierarchy of ILUC obtained in MIRAGE between vegetable oils and ethanol crops, and that the magnitude of ILUC factors obtained still seem reasonable given subsequent evidence. Using MIRAGE results does not require ignoring other work – rather, it would reflect a decision that the MIRAGE results remain relevant when considered in the context of other studies.

7.2.2 Rely on ILUC values from GLOBIOM work (Valin et al., 2015)

While the values from the GLOBIOM work are not reflected in the RED II itself, the GLOBIOM framework has been used by the European Commission more recently and informed the impact assessment on the RED II. It would therefore also be defensible for a Member State to take the GLOBIOM work as the basis for distinguishing between fuels at a regulatory level.

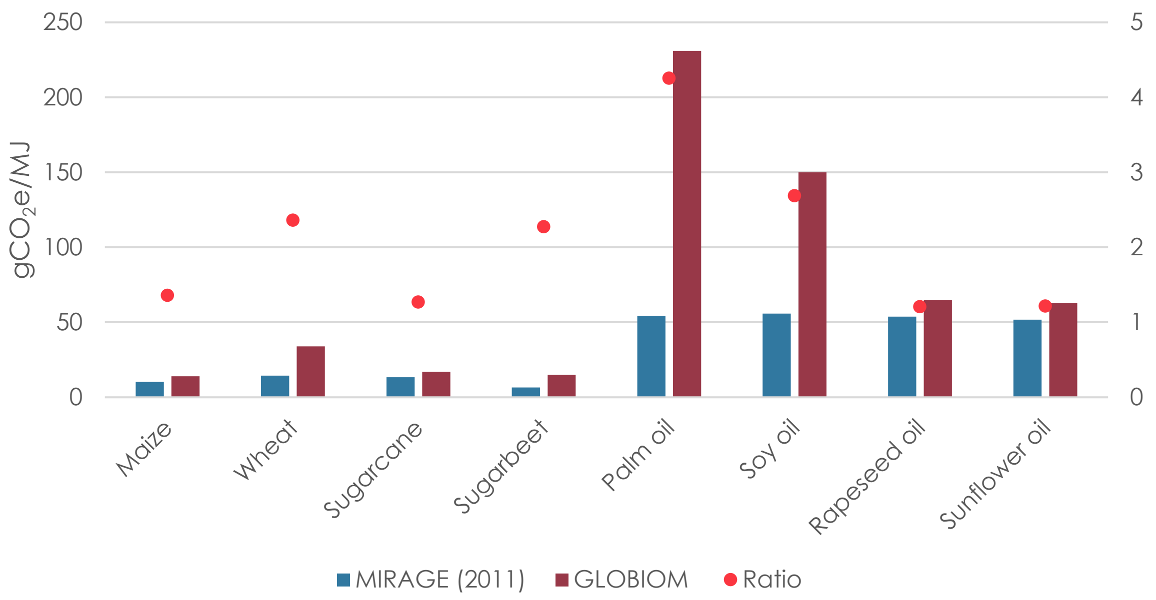

Figure 17 Comparing MIRAGE and GLOBIOM ILUC results. Right axis shows ratio of results, GLOBIOM:MIRAGE

Source: (Laborde, 2011; Valin et al., 2015)

As illustrated in Figure 17, the feedstock-specific ILUC numbers for food-based fuels from the GLOBIOM work are uniformly higher than those from MIRAGE. For rapeseed, sunflower, corn and sugarcane the difference is minor, and while the sugarbeet value is more than double the MIRAGE number it is still modest in absolute terms. For wheat, soybean oil and palm oil, however, the different is quite significant. From a regulatory point of view, the very high results (231 gCO_2e/MJ) for palm oil changes little because palm oil is already identified as high ILUC-risk. The main regulatory question if basing decisions on the GLOBIOM work is therefore whether wheat ethanol should receive less support than other food-based ethanol, and soy oil should receive less support than other non-palm vegetable oils.

7.2.3 Combine ILUC values from MIRAGE and GLOBIOM work

As an alternative to choosing to regulate based on one single set of ILUC results, a combined set of ILUC values could be created. The arithmetic and geometric mean values for the two studies are listed in Table 2.

Table 2 Arithmetic and geometric mean of the feedstock specific ILUC results from MIRGAE and GLOBIOM

	Corn	Wheat	Sugarcane	Sugarbeet	Palm oil	Soy oil	Rapeseed oil	Sunflower oil
Arithmetic mean	12	24	15	11	143	103	59	57
Geometric mean	12	22	15	10	112	91	59	57

Source: own calculation based on (Laborde, 2011; Valin et al., 2015)

The mean values do not change the hierarchy from the GLOBIOM results, but tend to reduce the differences between feedstocks compared to considering GLOBIOM on its own. Soy and palm oil would still have the highest values and ethanol feedstocks have lower values than the vegetable oils.

7.2.4 Determine ILUC values based on the broader set of results in Woltjer et al. (2017)

The review of the ILUC literature by Woltjer et al. (2017) was undertaken to support the European Commission's reporting obligations under the ILUC Directive. It states that, "Analysis of the best available scientific evidence was mainly focused on 30 studies that reported land use change (LUC) and indirect land use change (ILUC) factors". Woltjer et al. (2017) states that seventeen of these results were based on partial or general equilibrium economic modelling[32]. An eighteenth study, the current ILUC factors used in the California Low Carbon Fuel Standard regulation, was incorrectly described as based on expert opinion when it is also based on general equilibrium modelling (with GTAP). One of the studies identified as partial equilibrium based (Plevin et al., 2010) in fact presents a 'reduced form' spreadsheet model.

Five were based on what was referred to as 'empirical' approaches which were discussed in section 4.2, and one on causal descriptive modelling as discussed in section 4.1.

[32] The text suggests that some of these studies were from 'integrated assessment modelling' but in Table 9 of the review report all seventeen are identified as either partial or general equilibrium results (or both).

A further six studies are described by Woltjer et al. (2017) as 'hybrid LCA'. While these are classed as a group in the review, we would argue that this is not properly a separate category of evidence. Of these six: two (Acquaye et al., 2011, 2012) use ILUC factors based on work by Fritsche et al. (2010) (which is included separately as an empirical study); a third (Prapaspongsa & Gheewala, 2016) considers ILUC for biofuel consumption from cassava and molasses in Thailand and has limited application to the EU situation; a fourth (Boldrin & Astrup, 2015) refers to ILUC emissions estimates from a range of other equilibrium modelling exercises; a fifth (Bento & Klotz, 2014) presents results that are actually from general equilibrium modelling and the sixth uses partial equilibrium results from the U.S. EPA's analysis for the RFS. It is perhaps indicative of the difficulties of undertaking meta-analysis of ILUC modelling results that it is non-trivial even to come to agreement about what constitutes an original result!

Notwithstanding issues with the categorisation of some studies in Woltjer et al. (2017), it presents a fairly comprehensive review of ILUC work up to that time. Its summary of numerical ILUC results by feedstock is reproduced in Figure 18. The summary includes both mean and median values across the reports considered, but one should be cautious about the interpretation of these numbers. For example, we noted above that two of the included studies essentially reproduce the result from Fritsche et al. (2010), which therefore ends up being given extra weight in the statistical analysis. More generally, the data collection in Woltjer et al. (2017) aims to be comprehensive but in so doing takes a decision not to apply a filter based on an independent assessment of the quality of the results. For example, one could argue that the 'empirical' results reviewed by Woltjer et al. (2017) should be excluded from any calculation of a quantitative mean due to their limitations as non-consequential approaches to a consequential LCA question.

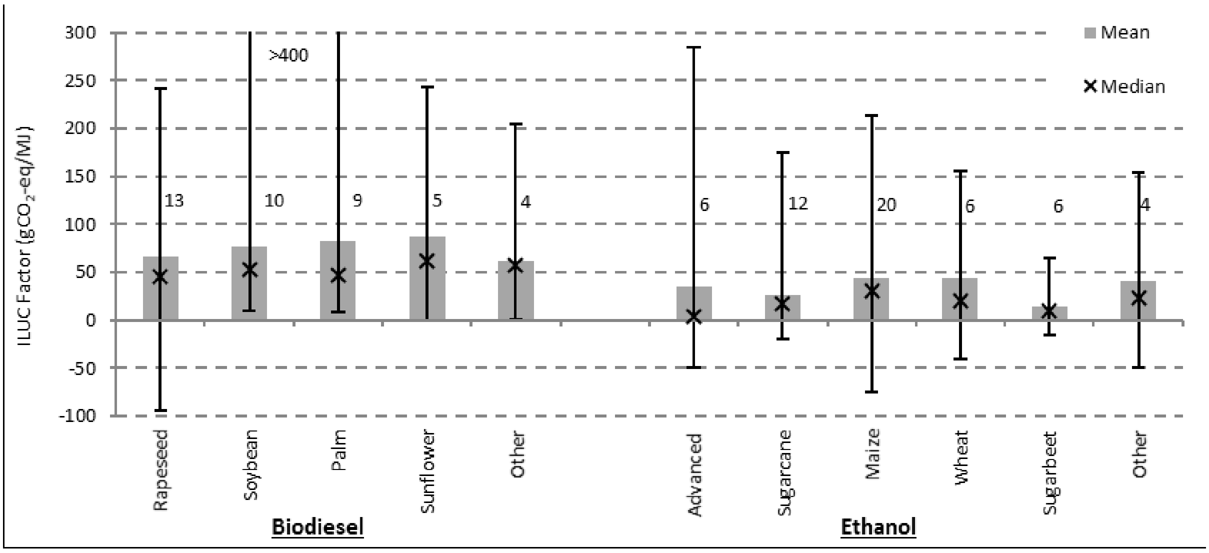

Figure 18 **ILUC mean values, median values and ranges presented by Woltjer et al. (2017)**

Filtering out as less relevant the studies that Woltjer et al. (2017) characterise as hybrid-LCA or empirical gives the mean ILUC values shown in Figure 19.

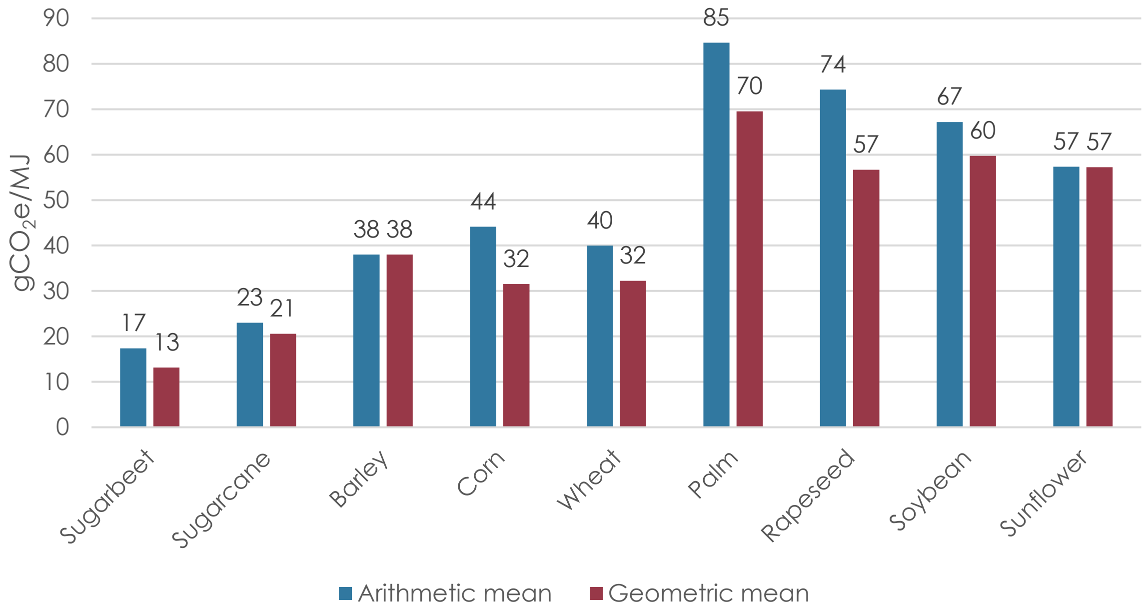

Figure 19 **Arithmetic and geometric mean of ILUC results from the most relevant studies on the list of best available scientific evidence from Woltjer et al. (2017)**

Note: excluding those identified as hybrid LCA or empirical, see text above.

Even having filtered out the studies described as hybrid LCA or empirical, one should be cautious about the interpretation of average values taken over a diverse set of papers in this way. Some models are represented several times, for example eight studies listed by Woltjer et al. (2017) are based on iterations of GTAP modelling. Averaging by study therefore gives greater weight for some feedstocks to GTAP than to other models. Averaging also gives equal weight to earlier as to later studies – arguably one should give greater weight to the most recent work (although even this cannot be taken for granted, as discussed in Malins et al., 2020). There is also an imbalance introduce between feedstocks by considering this larger set of results because some feedstocks like corn are considered in many studies, while others like sunflower are considered in only a few. Inclusion or exclusion of a feedstock from a modelling exercise with unusually high or low outcomes could therefore skew the relative results.

Notwithstanding the health warning on the interpretation of the average results shown in Figure 19, the hierarchy of ILUC emissions numbers remains similar to what is seen in the MIRAGE and GLOBIOM results. The biodiesel feedstocks have higher numbers than the ethanol feedstocks, and palm oil has the highest values of all. Across this set of studies sugars come out with lower values than cereals.

7.2.5 Indirect impacts from the use of wastes and residues

While the focus through much of the ILUC discussion has been on food crops, there is also potential for ILUC from energy crops (Pavlenko & Searle, 2018) and for indirect emissions associated with the use of materials thought of as wastes and residues, but which may already have some productive use (Malins, 2017b). These indirect emissions from waste use can include ILUC but may also include production emissions associated with replacement materials and combustion emissions associated with combustion of replacement fuels. As discussed in section 6.3, the discretion given to Member States to distinguish biofuel feedstocks based on ILUC emissions extends only to food and feed crops. Given this and the fact that many wastes and residues are listed in Annex IX of the RED II and therefore are explicitly identified as eligible for additional support. Denmark therefore may have limited legal space to further distinguish between waste and residual materials by expected indirect emissions.

8 Options for addressing ILUC in Danish regulation

Above, we have discussed some of the raft of analytical tools that have been used to investigate the ILUC associated with biofuel use and outlined the policy responses that have been taken by regulators in Europe and North America with a view to reflecting those concerns in decision making. The RED II provides EU Member States with a legal basis to differentiate support offered to biofuels on the basis of ILUC emissions, and in this section we review some policy options that could be available to the Danish government for reducing ILUC risk. We discuss what would be required to implement each option, how effective they might be in terms of reducing ILUC emissions, and potential associated administrative costs and challenges.

Any regulatory action to distinguish between biofuels based on their ILUC characteristics must be based on scientific evidence and must not be introduced in a way that unfairly favours domestic producers of feedstock. Policy action that failed to follow this principle would be vulnerable to challenge through the World Trade Organisation. In practice, this means that it is preferable to explicitly regulate based on a specified principle than to simply identify certain products as being subject to greater restrictions. The high ILUC-risk analysis provides an example of this – while in practice the mechanism leads to restrictions on the use of palm oil, this outcome is achieved through a well justified data-based standard (the threshold value for high carbon stock land conversion) rather than simply by adding palm oil to a list of restricted feedstocks.

Currently, we understand that Denmark has an overall biofuel blending obligation (iblandingsforpligtelsen) and an advanced biofuel blending obligation (avancerede iblandingsforpligtelse), both framed in energy terms (Forsyningsministeriet, 2020). It also has a GHG emissions reduction obligation to implement Article 7a of the FQD (reduktionsforpligtelsen), which requires that from 2020 onwards the average GHG intensity of the fuel supplied should be at least 6% below the 2010 GHG intensity baseline. There is an expectation that from 2024 a more stringent GHG reduction obligation will become the main mechanism supporting biofuel use in Denmark, and that the blending targets will be retired. Shifting to a solely GHG based system would be consistent with the proposed amendment to the RED II that is currently being reviewed by the European institutions.

These requirements are to be fulfilled by surrendering appropriate numbers of tradable 'biotickets' which demonstrate the supply of volumes of relevant fuels. In the discussion in this section, we use the term biotickets to refer to compliance credits generically – for several of the options discussed, this would involve revisions to the current bioticket system.

8.1 Through the use of additional 'caps'

As noted in section 6.3, Article 26(1) of the RED II gives as an example of ILUC-based regulation the possibility of setting a lower cap on the contribution to targets of biofuels from oil crops than biofuels other food or feed resources (starch and sugar crops). Oil crops are identified because the ILUC factor given for oil crops in Annex VIII of the RED II is 55 gCO_2e/MJ, significantly higher than the values given for starch crops and sugar crops (12 gCO_2e/MJ and 13 gCO_2e/MJ respectively). Not only is the value assigned to oil crops significantly higher in absolute terms, but it is also large compared to the calculated direct

GHG savings values required for biofuels supplied under the RED II. To qualify for support under the RED, an oil-crop-based biodiesel may have a maximum of 47 gCO_2e/MJ reportable direct emissions (for a facility operational prior to 5 October 2015). Adding the Annex VIII ILUC value would give a hybrid LCA emission factor of 102 gCO_2e/MJ, higher than the fossil fuel comparator.

Additional limits could be implemented within the overall cap on food-based fuels targeting the feedstocks with the highest expected ILUC emissions. For example, if the overall cap on food-based fuels was set at 7% of transport energy, the contribution from food-based biodiesel could be capped at 2 or 3%, and the contribution from palm oil and soy oil at 1% (as palm oil use is to be reduced as 'high-ILUC risk' anyway this would primarily represent a limit on increased soy oil use).

8.1.1 Compatibility with the RED II

The RED II explicitly identifies the imposition of additional limits on vegetable oil use as a legitimate use of Article 26(1). It is not so explicit whether limits on specific individual feedstocks such as soy oil would be permissible, but our reading of Article 26(1) is that such measures would also be acceptable if clearly justified based on the available evidence base.

8.1.2 Administrative burden

Imposing additional caps on feedstock use would require only limited changes to existing systems for tracking and redeeming biotickets. We would therefore not expect significant additional burden on the national authorities from implementing such measures.

8.2 Hybrid LCA with GHG-based fuel supplier targets

The targets for renewable energy in transport in the RED II are framed in energy terms, but some Member States such as Germany have implemented support systems where levels of support are based on reportable GHG intensity (similar to the California LCFS system). Under the Fit for 55 package a revision to the RED II has been proposed that would move the European-level requirements from an energy basis to a GHG basis. Under GHG-based support systems the contribution of a batch of fuel towards fuel supplier obligations is proportional to the calculated GHG saving delivered by that fuel (the amount of energy supplied multiplied by the difference between the fossil fuel comparator GHG intensity and the reportable GHG intensity of the fuel batch). This calculation allows compliance credit to be expressed in terms of tonnes of CO_2 saved.

Currently, the EU Member States that apply GHG-based support systems base the level of support given on the GHG intensity score as calculated under the RED LCA methodology, but Denmark could in principle move to a GHG-based system of crediting for alternative fuels including ILUC emissions in the LCA requirement (requiring fuel suppliers to report their GHG intensity on a hybrid LCA basis rather than based only on the RED LCA methodology).

One advantage of introducing ILUC factors as part of a hybrid-LCA based performance metric is that it would help to put support for renewable fuels on a more technology neutral basis again. As discussed in the introduction, when GHG-based support systems do not include any characterisation of indirect emissions this can be seen as distorting the level playing field between technologies.

The hybrid LCA requirement need not also be applied for assessment of compliance with the minimum GHG saving thresholds under Article 29 of RED. Batches of fuel would be eligible for support if the GHG saving on the RED methodology met the GHG saving thresholds in the Directive, but the level of support given would be determined by the hybrid LCA score. This is illustrated in Table 3 using some example fuel pathways and the ILUC values from RED II Annex VIII.

Table 3 Example of credit generation by various fuels under a GHG crediting system with hybrid LCA

Example fuel	Date of first operation	Article 29 GHG saving threshold	Reportable GHG intensity under RED II	ILUC factor (Annex VIII)	Reportable saving under RED II	Reportable saving under hybrid LCA	Credit per 1000 litres (tCO_2e)
Soy oil biodiesel	30-Sep-14	50%	29	55	69.10%	10.60%	0.3
Rapeseed oil biodiesel	18-Nov-16	60%	34	55	63.80%	5.30%	0.2
UCO biodiesel	02-Apr-20	60%	14	0	85.10%	85.10%	2.6
Corn ethanol	23-Mar-14	50%	41	12	56.40%	43.60%	1.1
Corn ethanol	04-Oct-13	50%	29	12	69.10%	56.40%	1.4
Sugar beet ethanol	12-Mar-11	50%	38	13	59.60%	45.70%	1.2
Food waste ethanol	07-Jan-19	60%	29	0	69.10%	69.10%	1.8

Pathway GHG intensities based on examples taken from UK Department for Transport (2019), dates of first operation are dummy data used only to illustrate the interaction with the GHG saving threshold in Article 29. Pink highlights show pathways that would not meet the relevant GHG thresholds with ILUC included. Yellow highlights show which pathways would receive the most credit under the hybrid LCA approach.

It can be seen in Table 3 that the inclusion of ILUC values in a GHG crediting system could very significantly change the value proposition for food-based biofuels. Consider the rapeseed oil biodiesel and UCO biodiesel pathways in the table. Under the current Danish biofuel support system, the UCO biodiesel would be 'double counted' due to being listed as a waste feedstock in Part B of Annex IX of the RED, and therefore receive twice as much support per unit volume as the rapeseed biodiesel. Under a GHG crediting system excluding ILUC emissions, this difference would be narrowed. The UCO biodiesel in the example delivers 33% more GHG reduction than the rapeseed biodiesel, and therefore would get a third more support. Moving to a hybrid LCA basis, however, would result in the UCO biodiesel delivering thirteen times more reportable GHG reductions than the rapeseed biodiesel, and therefore receiving thirteen times more support. This large multiplier reflects the very limited GHG benefit calculated for the rapeseed biodiesel under the hybrid LCA system. This would provide a very firm signal in favour of a switch from food-oil feedstocks to alternative biodiesel and renewable diesel feedstocks.

This approach could also be expected to significantly advantage food-based ethanol over food-based biodiesel due to the difference in estimated ILUC emissions. The sugarbeet ethanol pathway in the example would receive six times as much support as the rapeseed biodiesel pathway. Introducing this value differential between food-based ethanol and biodiesel could be expected to push up consumption of ethanol, although this shift would be restricted by blend limitations.

Within the ethanol market, the differentiation between food-based and other feedstocks under a hybrid LCA approach would be much less marked. In Table 3, consider the example pathways for ethanol from food waste and ethanol from sugar beets. If we assume that the food waste is gathered from industrial sources and is not fit for feed use, and therefore is consistent with the definition of the biomass fraction of industrial waste given in Annex IX Part A (d), then food waste ethanol would currently be double counted and could be counted towards advanced biofuel targets. This fuel would therefore receive at least double the support available to food-based ethanol. Under a GHG-based system on the RED methodology, however, it would only receive 16% more support than the sugar beet ethanol pathway (not considering support from any sub-targets). Under the hybrid LCA scheme, the food-waste ethanol pathway would receive 50% more support than sugar beet ethanol. We see then that in the ethanol market a GHG-based system with a hybrid-LCA would reduce the value difference between crop-based and waste-based fuels, unless coupled to an advanced biofuel sub-target.

8.2.1 Compatibility with the RED II

In section 6.3, we have argued that Article 26(1) provides a broad flexibility for Member States to adjust the level of financial support available to biofuels from food and feed crops informed by best available evidence on ILUC emissions. The use of a hybrid LCA value to set the level of support clearly implies a more complex regulatory framing than simply limiting the quantity of food-oil based fuels that can be supported, but we believe that it is consistent with the Directive as written. An argument against allowing a hybrid LCA system can be found in Recital 81 of the RED II, which states that,

> *"The level of greenhouse gas emissions caused by indirect land-use change cannot be unequivocally determined with the level of precision required to be included in the greenhouse gas emission calculation methodology."*

This recital is included to provide context for the decision in the RED II to manage ILUC risk via the use of the food-cap and the high ILUC-risk designation, but it does clearly signal a concern from the European Institutions about giving ILUC factors direct regulatory application. The European Commission can be expected to give considerable weight to this recital and would be likely to challenge any Member State proposing to introduce ILUC factors as part of a GHG-based support system. As with other policy options, the Commission could also be expected to be more cautious of a hybrid approach based on ILUC estimates from new bespoke research for the Danish Government (treated as 'best available evidence', cf. chapter 7) than if such an approach relied on the values already included in Annex VIII of the RED II, or directly on the MIRAGE and GLOBIOM modelling work.

The recitals in European Directives have a primarily interpretive role (Baratta, 2014), and cannot create additional legal obligations on Member States that are not present in the articles of a Directive, but they can be invoked in the case of legal dispute as indicative of the intention of the articles. While this recital does clearly present an argument against the use of ILUC factors in the LCA requirements set by the RED II, a clear legal argument could be made that Article 26(1) takes precedence. Article 26(1) clearly gives Member States the prerogative to take estimated ILUC impacts into account in ways that go beyond the requirements for a food cap and the phase out of high ILUC-risk fuels. Recital 81 confirms that the European Institutions have determined that the use of ILUC factors in the GHG calculation should not be imposed on Member States, but it is our opinion that it should not be interpreted as constraining the scope for Member State action under Article 26(1).

If hybrid LCA were to be used in this way, one important implication would be that the sum of the GHG savings delivered by any given fuel supplier as assessed under the hybrid LCA system would not be equal to the sum of the GHG savings delivered as calculated by the RED methodology (more specifically, if any food-based fuels with positive ILUC factors are used, then the sum of GHG savings calculated under the hybrid LCA approach would always be less than the sum as calculated under the RED methodology). There is precedent under the first RED for Member States setting targets for fuel suppliers on a different basis than the energy targets in the Directive. Some Member States have set targets based on volume of fuel supplied – when converted to energy terms, this results in ethanol being supported more generously than biodiesel. Other Member States have set GHG reduction targets, although to date these have not included ILUC factors. In these cases where fuel supplier targets are set in units other than energy supplied, the Member States were required to set them at a level that could reasonably be expected to deliver compliance with the renewable energy use target set in Article 3 of the RED I. Similarly, if Denmark were to adopt a hybrid-LCA-based support system it would need to set fuel supplier GHG intensity reduction requirements that would be consistent with achieving the RED II targets (whether the current target for 14% renewable energy in transport or a revised target under the Fit for 55 package).

8.2.2 Administrative burden

Introducing a hybrid LCA system would represent a greater additional commitment for both Danish regulators and Danish fuel suppliers than simply imposing limits on vegetable oil use. Firstly, the DEA would need to identify, propose and justify a set of ILUC factors to be used. Unless simply using the values already included in Annex VIII of the RED II this would be likely to require some form of consultative engagement with the stakeholder community. Secondly, some form of impact analysis would be necessary in deciding the appropriate level for a GHG target under a hybrid LCA system.

Once the basic structure of the obligation was determined, ILUC factors could be added to a GHG-based accounting system relatively easily by adding an additional term to the LCA requirements. Fuel suppliers should be able to take advantage of experience gained in delivering compliance with the 2020 FQD 7a target, but may still require operational support to deal with the slightly more complex LCA requirement. Excluding any resources invested in developing ILUC estimates, we would expect that moving to a hybrid LCA-based system of GHG targets would impose at most modest overheads on the regulator.

8.3 Including ILUC emissions in assessment against minimum saving thresholds

An alternative use of a hybrid LCA approach would be to use a hybrid LCA value in assessing compliance with the minimum GHG reduction thresholds of the RED. This could be done in the context of either energy-based or GHG-based support. This would place a tighter limit on the range of biofuels that could receive support – looking again at the example pathways in Table 3, if sustainability compliance were assessed based on the hybrid LCA results instead, only one of the food-based biofuel pathways (the second corn ethanol pathway) given in the table would still meet the threshold.

Given a high ILUC value on oil crops (anything more than perhaps 40 gCO_2e/MJ), including ILUC emissions in the assessment against minimum GHG saving thresholds would have the practical effect of blocking essentially all support to crop-based biodiesel. With the 55 gCO_2e/MJ ILUC emissions listed in Annex VIII of the RED II, an oil-crop would need to report negative emissions on the direct part of the LCA in order to meet the threshold – this would only be possible if reporting biomass carbon or soil carbon accumulation at the farm producing the feedstock. For starchy and sugary crops, compliance with the sustainability requirement would be more achievable, but still difficult for many currently compliant pathways (depending on the precise ILUC values adopted, of course). Under the Annex VIII ILUC values, a fuel production pathway that currently just meets the threshold would have to improve its direct GHG intensity by up to 13 gCO_2e/MJ. This may be possible through measures such as fertiliser use optimisation or moving to renewable process energy.

8.3.1 Compatibility with the RED II

The inclusion of ILUC factors on a hybrid LCA basis when assessing compliance with the GHG saving thresholds in Article 29(10) is not directly considered in the RED. Article 26(1) refers to distinguishing on the basis of ILUC, for the purposes of Article 29(1). Article 29(1) in turn refers to the GHG thresholds in Article 29(10) as a requisite for biofuels to be counted towards the Union target and Member State and fuel supplier obligations, or to be eligible for financial support. While this link from Article 26(1) could be invoked to support the inclusion of ILUC factors when assessing biofuels against the Article 29(10) thresholds, Article 31(1) lays down the rules for the calculation of GHG savings for the purpose of Article 29(10) and explicitly requires the use of the methodology laid out in Annexes V and VI of the RED II. Given that these annexes do not make any accommodation for the consideration of ILUC emissions, the European Commission may determine that Member States do not have the flexibility to include ILUC emissions when assessing compliance of a biofuel with the Article 29(10) thresholds.

8.3.2 Administrative burden

Adding ILUC factors to the assessment of compliance with minimum GHG saving thresholds would be relatively administratively simple, in the sense that it would be easy to cross reference the reported GHG intensity of each fuel batch and its feedstock against the list of ILUC factors to determine compliance. The main administrative challenge we would foresee with implementing this system would be that Denmark would have a different eligibility standard than other EU Member States, and therefore could no longer treat certification by approved voluntary schemes as meeting the EU GHG thresholds as adequate on its own to demonstrate compliance with the Danish standard. This may create an additional administrative burden confirming the results of threshold calculations.

8.4 Extension of the high ILUC-risk concept

As noted in section 4.2, The RED II introduced the concept of high ILUC-risk biofuels. The idea of the high ILUC-risk mechanism is that while ILUC assessment in general is plagued by uncertainty, it is reasonable to take steps to reduce to support for the use of crops that are directly associated with loss of high carbon stock land. The RED II requires that any biofuel feedstocks for which more than 10% of global expansion[33] is identified as occurring at the expense of high carbon stock areas shall be characterised as 'high ILUC-risk' and that those fuels should be made ineligible to count towards targets and for support under Member State schemes by 2030. Based on the first ILUC-risk assessment by the European Commission, only palm oil is currently identified as high ILUC-risk (European Commission, 2019). The results of this assessment are shown in Table 4.

Table 4 Share of production of biofuel crops identified by the European Commission as coming at the expense of high carbon stock land

Crop	Share of expansion on forested land	Share of expansion on peatland	Productivity factor	ILUC risk score (productivity and carbon adjusted share of expansion on high carbon stock land)
Wheat	1%		1	1%
Corn	4%		1.7	2%
Sugar cane	5%		2.2	2%
Sugar beet	0%		3.2	0%
Rapeseed	1%		1	1%
Palm oil	45%	23%	2.5	42%
Soybean	8%		1	8%
Sunflower	1%		1	1%

Source: European Union (2019)

It can be seen that palm oil has by some distance the highest 'ILUC risk score' at 42%. This is well above the 10% threshold value. The European Commission is expected to periodically review the assessment, but it would take a dramatic and demonstrable

[33] The score is in fact adjusted for productivity, so that as a more productive crop the threshold for palm oil is effectively set at 25% expansion into forestland. Peatland is also counted 2.6 times in the calculation in recognition of its higher carbon stocks compared to forest areas. We henceforth refer to the productivity and peat-carbon adjusted share of expansion into high carbon stock areas as the 'ILUC risk score'. There is also a minimum requirement on total rate of expansion before a crop can be considered high ILUC-risk.

reduction in the rate of deforestation and peat loss associated with palm oil for the classification to be revised. The second highest ILUC risk score is soy oil, 8%. This falls below the 10% threshold, but it is plausible that a reassessment might result in the classification of soy oil being changed if the data shows an uptick in deforestation rates (cf. Malins, 2020). The other crops assessed all have relatively low ILUC risk scores of 2% or less, and it would seem less likely that any of those would be reassessed into the high ILUC-risk category unless very significant changes in expansion patterns are observed.

Denmark could consider building on the high ILUC-risk principle by imposing additional restrictions on biofuel feedstocks with an intermediate ILUC-risk score based on the current ILUC risk assessment. Setting an intermediate category in the range 5-10% would currently affect only soy oil, though this could change with the Commission's next report. Setting the intermediate threshold lower, for example at 2%, could bring additional feedstock into the category (corn and sugar cane based on the current assessment). Doing so could be controversial, as whereas for palm and soy oil the results of the high ILUC-risk assessment are consistent with the results of ILUC modelling corn and sugar cane tend to have relatively low ILUC estimates in modelling. An ILUC risk score of 2% could also be seen as reflecting a fairly low risk of deforestation – for those feedstocks land use change emissions from non-forest land conversions might be expected to be an equal or greater emissions source, making it less justifiable to focus only on high carbon stock land. While the RED II does not make any requirement that the high ILUC-risk assessments should align with ILUC model results, we would suggest that the ILUC-risk approach is on firmer ground when the resulting regulatory action is also supported by best evidence from ILUC models.

The most obvious ways to restrict support to 'intermediate ILUC-risk' fuels would be to impose a limit on total supply volume (to be phased in on a similar schedule to the restriction on high ILUC-risk biofuels) or to create a category of lower value biotickets, for example by counting intermediate ILUC-risk fuels as half of their physical energy content.

8.4.1 Market mediated ILUC-risk

The high ILUC-risk framework is intended as a way to use a direct impact metric to inform regulatory decisions in order to reduce expected indirect impacts. It therefore differs from ILUC analysis in that it ignores market linkages between feedstocks. This is particularly important among vegetable oil feedstocks, as the lack of market linkages results in rapeseed and sunflower oils being given very low ILUC-risk scores even though they are linked by the vegetable oil market to high ILUC-risk palm oil. It would be possible to reintroduce an element of consequential thinking to the high ILUC-risk approach by considering that demand for one vegetable oil may trigger supply of another. Figure 20 and Figure 21 show results from MIRAGE and GLOBIOM respectively for the change in vegetable oil supply associated with increase in demand for biodiesel from a single feedstock. On the left of each chart, we see that both models assume that when palm oil demand increases this is overwhelmingly met by increased palm oil production. In contrast, the third bar of each chart shows that a large fraction of an increase in soy oil demand is expected to be met with increased palm oil production. This difference is explained by the fact that soy oil represents less than half of the value of the soy crop whereas palm oil is most of the value of the palm crop. We therefore expect palm oil supply to be more sensitive to vegetable oil demand than soy oil supply is.

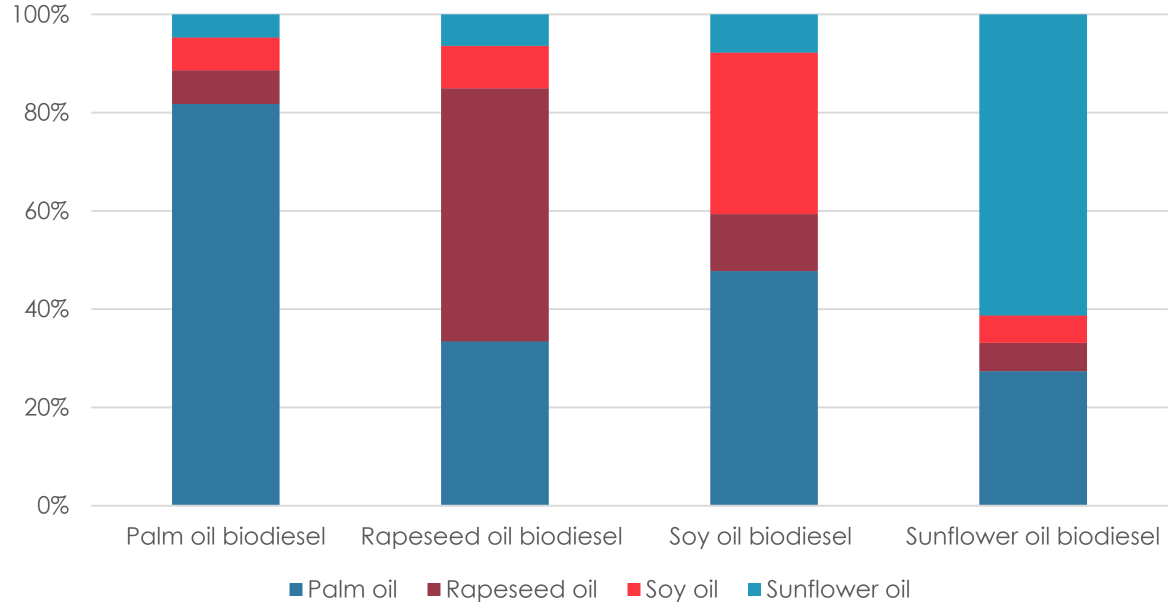

Figure 20 **Increases in vegetable oil production in response to demand for different biodiesel feedstocks in MIRAGE**

Source: Laborde (2011)

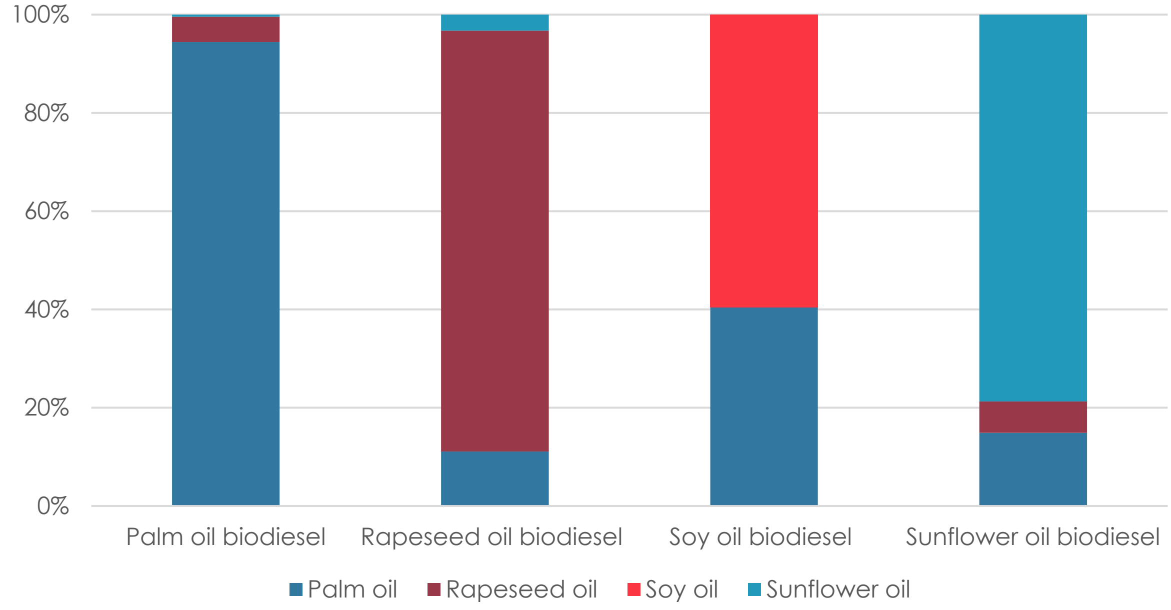

Figure 21 **Increases in vegetable oil production in response to demand for different biodiesel feedstocks in GLOBIOM**

Source: Valin et al. (2015)

By combining the ILUC-risk scores from the European Commission analysis with the ILUC model outputs on the amount of additional consumption met by increased supply of each oil, it is possible to calculate a new 'market mediated ILUC-risk' metric.

Table 5 ILUC-risk scores obtained by cross referencing feedstock ILUC-risk assessments with model data on combination of oils expected to meet additional demand

	'Market mediated' ILUC-risk score
Palm oil biodiesel	37.2%
Rapeseed oil biodiesel	10.5%
Soy oil biodiesel	22.3%
Sunflower oil biodiesel	10.0%

Source: Own calculation based on (European Commission, 2019; Laborde, 2011; Valin et al., 2015)

The resulting metric shows the hierarchy that one would expect – the strongest deforestation link is for palm oil, then soy oil and then rapeseed and sunflower oil. Both rapeseed and sunflower have scores on this new metric of 10% or more. Some form of limitation could be imposed on feedstocks that have a market mediated ILUC risk above 10% or some other threshold, which would provide an alternative (or complementary) justification for limits on these other vegetable oils.

The obvious criticism of such a market-mediated ILUC-risk metric would be that if one goes so far as combining outputs from economic models with data on deforestation and peat loss risk, why not go the full distance and take the ILUC results from the model as the main regulatory metric? There is no simple answer to that challenge, and such a compound metric would be likely to be perceived as complex (perhaps over-complex) by stakeholders. Nevertheless, results such as this provide a useful reminder that the high ILUC-risk framework as it stands fails to capture the full risk of ILUC emissions associated with the other vegetable oils.

8.4.2 Compatibility with the RED II

The RED II defines only the high ILUC-risk category, and therefore adding a 'intermediate ILUC-risk' category would need to be justified under Article 26(1) as a form of additional limit on biofuel supply. In practice, adding a new intermediate ILUC-risk category would be comparable in legal terms to imposing additional limits on a single feedstock as discussed in section 8.1.1, and we believe it would be defensible in the terms of Article 26(1).

The market-mediated ILUC-risk calculation presented above is novel, and combines elements of the high ILUC-risk assessment with ILUC modelling. Given that the intention of the high ILUC-risk assessment in the RED II is to allow measures to reduce ILUC to be defined without relying on ILUC modelling results it might be expected that the European Commission would be reluctant to endorse such an approach.

8.4.3 Administrative burden

Creating an additional category of intermediate ILUC-risk crops would require some initial setup within the Danish system, but (much as discussed in section 8.1.2) we would not anticipate a significant longer-term increase in administrative burden.

8.5 Extension of the low ILUC-risk concept

Alongside the high ILUC-risk category of biofuels created in the RED II is the low ILUC-risk concept. In the RED II, low ILUC-risk fuels are defined as biofuels from food- or feed-crops that are produced in ways that avoid indirect land use change by either increasing productivity or by cultivating areas that would otherwise not be productively farmed. Central to the premise of low ILUC-risk feedstock production is the idea that biofuel producers can support the production of 'additional' feedstock that would not otherwise have been produced, and thereby avoid impacting existing commodity markets. The idea of avoiding ILUC emissions by driving additional feedstock production has been discussed for as long as ILUC has been a concern, forming a central recommendation of the UK's Gallagher review, for example RFA (2008).

The certification of low ILUC-risk fuels is a complicated area and the European Commission is in the process of further developing certification requirements (European Commission, 2021a). As noted in Malins (2019a), to be effective a low ILUC-risk certification system must ensure that low ILUC-risk projects are truly delivering additional feedstock production driven by the associated biofuel mandate. Simply certifying feedstock material from projects that would have happened anyway will do little or nothing to reduce global land use change emissions, even if those projects have positive characteristics when considered in isolation. It is beyond the scope of this report to fully reiterate the challenges involved in developing an effective system of low ILUC-risk certification. In this section we develop ideas for the expansion of the role of low ILUC-risk certification on the assumption that the certification can be implemented in a way that is effective.

European Union (2019) defines additionality measures as, "any improvement of agricultural practices leading, in a sustainable manner, to an **increase in yields of food and feed crops** on land that is already used for the cultivation of food and feed crops; and any action that enables the **cultivation of food and feed crops on unused land**, including abandoned land, for the production of biofuels" (our emphasis). It is required that producers should demonstrate that projects, "become financially attractive or face no barrier preventing their implementation only because the biofuels … produced from the additional feedstock can be counted towards the targets for renewable energy".

Because low ILUC-risk biofuel production requires delivering additional production, the largest opportunities for low ILUC-risk biofuel production are identified in regions where there are currently either significant inefficiencies in agricultural production or large areas of available land that could be converted to agriculture with low carbon cost. Dehue et al. (2010) notes that the potential to increase productivity will generally be greater in countries with less developed agricultural systems where there are large gaps between current and achievable yields. For example, Brinkman et al. (2021) discusses the potential for producing low ILUC-risk rapeseed in Eastern Romania by closing the gap between local average yields and EU average yields. Better mechanisation, better fertilisation and increased use of pesticides are identified as actions that could be take. In a country such as Denmark which already delivers a higher average yield for rapeseed than is achieved in most other EU Member States[34] it may be difficult to deliver and demonstrate further productivity gains. Similarly, opportunities to bring unused land into production in Denmark may be rather more limited than in larger countries with less developed agricultural sectors. Supplying low ILUC-risk biofuels would therefore be likely to require the use of imported feedstocks, even for feedstocks such as rapeseed that are grown in Denmark.

An upside to introducing a low ILUC-risk biofuel support scheme is that as well as avoiding ILUC emissions on the certified batches, there is the potential to catalyse longer term improvements in agricultural systems in the source regions. There is potential for productivity increase measures demonstrated in the context of low ILUC risk projects to then be more broadly adopted. Similarly, demonstrating the successful introduction or return to agriculture of degraded or abandoned land through low ILUC risk schemes could encourage similar actions in the broader supply chain. Deploying such measures more widely would have the potential to increase food production and deliver increased incomes to rural populations in the areas where these systems would be developed.

In the RED II as it stands, low ILUC-risk certification fulfils a well-defined but limited regulatory role – it provides a basis for feedstocks that are generally identified as high ILUC-risk to be used in the EU in 2030 if they are from low ILUC-risk certified projects. Currently, this means that low ILUC-risk-certified palm oil has a defined market opportunity under the RED II but there is no direct value proposition provided for other low ILUC-risk feedstocks such as rapeseed or sunflower. As an alternative or complement to imposing additional limitations on fuels believed to be associated with significant ILUC emissions, Denmark could introduce additional incentives to support the supply of low ILUC-risk fuels. Before making any final regulatory decisions, it would be appropriate to wait for finalisation of the draft implementing regulation on rules to verify sustainability and greenhouse gas emissions saving criteria and low indirect land-use change-risk criteria and publication of the Commission's forthcoming reports on low ILUC-risk guidance and low ILUC-risk pilots.

8.5.1 Multiple counting for low ILUC-risk fuels

A simple option to support low ILUC-risk biofuels would be to make certified fuels eligible for an increased number of biotickets, for example awarding 1.5 times the number of biotickets per unit of energy supplied. A multiple counting entitlement in Denmark would, to the best of our knowledge, be the strongest support currently available to low ILUC-risk systems anywhere in the EU and could therefore be a significant contribution to actualising these production models.

8.5.2 Exemption from limits on the use of food-based fuels

In the RED II, low ILUC-risk biofuels from food crops are subject to the overall cap on the use of food-based fuels (unless the low ILUC-risk fuels could also be characterised as intermediate crops, which are outside of the RED II definition of food and feed). They are exempted from the restrictions on the use of high ILUC-risk fuels.

If introducing additional limits on the use of food-oils for biodiesel, certified low ILUC-risk fuels could also be exempted from those limits. Certified fuels would then be able

[34] Based on Eurostat data https://ec.europa.eu/eurostat/databrowser/bookmark/857c068a-34bd-4ec8-94c2-a8ec075d0e55?lang=en

to compete for market space with first generation ethanol and with advanced and waste-based biofuels.

8.5.3 Coupling the requirement for low ILUC-risk fuels to the supply of food-based fuels

An alternative mechanism to drive production of low ILUC-risk feedstock could be to require a certain number of low ILUC-risk biotickets to be redeemed when biotickets from 'standard' food-based biofuels are used to meet renewable fuel supply obligations. For example, a requirement could be introduced that by 2030 for every ten biotickets redeemed for food-based biofuels one bioticket must be redeemed for low ILUC-risk biofuels.

A requirement of this sort, tying the development of low ILUC-risk feedstock production systems to the level of continued use of food-based biofuels, would give the market a choice between shifting more completely to the use of non-food feedstocks or developing low ILUC-risk supply chains as a growing fraction of the feedstock pool for first generation fuels. A potential downside of this form of support would be that a fuel supplier could invest in good faith in developing low ILUC-risk systems to meet their obligations, only to find that adverse conditions outside their control led to lower-than-expected production of certified material in a given year (in a year with a bad harvest due to exogenous factors such as poor weather, it would be perfectly possible for a well-conceived productivity project to produce no certifiable material). In such a situation, a supplier may be forced to choose between being out of compliance with the requirement to supply low ILUC-risk fuels or dramatically scaling back supply of food-based fuels and potentially failing to comply with the main targets. Such issues would be far less acute under a multiple counting approach, as there would be no minimum supply requirement and a shortfall in low ILUC-risk feedstock supply could be compensated with other fuel types.

8.5.4 Compatibility with the RED II

The low ILUC-risk concept is defined within the RED II (and associated delegated acts by the European Commission), and therefore the use of European Commission approved low ILUC-risk certification systems as a basis to provide increased support to some biofuels would be expected to be recognised by the Commission as based on the best available evidence on ILUC. The example of Member State action under Article 26(1) of the RED II (imposing limits on the use of crop-based biofuels) is an essentially negative measure, whereas providing additional support to low ILUC-risk fuels would be a positive measure. Nothing in the text of Article 26(1), however, appears to preclude the use of measures to distinguish between biofuels through added support instead of by tighter limitations. Recital 91 of the RED II states that, "Feedstock which has low indirect land-use change impacts when used for biofuels, should be promoted for its contribution to the decarbonisation of the economy." While this does not single out low ILUC-risk production systems as such, it supports Member States in taking additional action within the scope of Article 26(1) to promote any feedstocks with low ILUC impacts. This said, it should be noted that when the RED II was passed it would have been possible for low ILUC-risk fuels to be given additional support by inclusion on Annex IX or by some similar measure. We do not believe, however, that the lack of such support at EU level should be taken to imply a prohibition on action by Member States.

8.5.5 Administrative burden

Introducing additional support for low ILUC-risk fuels would require systematic engagement with certification systems that are still under development, and therefore could involve a greater administrative burden than some of the other options discussed. We would expect that after the low ILUC-risk requirements are finalised there will be Commission approved voluntary schemes available offering certification of low ILUC-risk feedstock. It would be relatively administratively simple to offer enhanced support to suppliers able to provide evidence of such certification.

Given that we do not expect most Member States to introduce additional support measures for low ILUC-risk fuels, it should be recognised that Denmark could be the only EU Member State in which certification of feedstock other than palm oil had a defined value. The Commission is likely to be focused on the quality of low ILUC-risk certification for high ILUC risk fuels. In this context, the Danish Government might consider it necessary to engage with certification bodies to provide additional assurance that approaches for certifying other feedstocks were fit for purpose, which would imply additional staff resources.

For Danish fuel suppliers, developing relationships with low ILUC-risk feedstock producers would represent an additional burden also requiring commitment of staff resources (or offering funding to appropriate consultants). Under a multiple counting scheme, engagement on low ILUC-certification would be entirely optional and therefore economic operators would be free to decide whether to engage. Under the 'coupled mandate' approach outlined in section 8.5.2, fuel suppliers would need to supply some low ILUC-risk fuel in order to continue supplying other food-based fuels, giving them less flexibility to decide whether to participate. This might therefore be seen as a more burdensome regulatory approach.

8.6 Developing risk ratings for a broader set of externalities

While issues such as the 'food versus fuel' debate and the impact of deforestation on biodiversity have featured in the European biofuel discourse, the focus when discussing indirect land use change has always been the associated GHG emissions. The reasons for this focus are clear – the RED is explicitly identified as part of the EU's package of climate change mitigation policies, and if there are biofuels being supplied under the RED that result in net increases instead of reductions in global GHG emissions, that would represent a clear policy failure. It is more difficult to balance the interaction between biofuel demand and other social and environmental indicators – even if we accept that biofuel mandates will tend to increase food prices and undermine food security to some extent (Malins, 2017a), it is not obvious what should constitute too great an impact. At present, the EU has come to a compromise between ambition in renewable fuel targets and concern about ILUC and food markets by limiting the support for consumption of food-based fuels. Similarly, while most analytical attention has been placed on the GHG emissions associated with land use changes, increased agricultural production can also threaten biodiversity. The RED II places limits on the direct conversion of biodiverse systems for biofuel production, but does nothing to manage any indirect impacts. In some cases, biodiversity impact and carbon impact are somewhat correlated, such as when biofuel demand is found to have driven tropical deforestation. In other cases, however, negative biodiversity impacts might be expected without a significant accompanying reduction in carbon stocks. It might, for example, be possible to expand palm oil production in some biodiverse tropical grassland and increase total carbon storage per hectare through the carbon in the palm trees themselves, while still undermining biodiversity goals.

To date, policy makers have not integrated consideration of these other externalities from biofuel production when differentiating between feedstocks. One reason for this may be that it is not at all simple to develop metrics for such concerns. As discussed in some detail in the sections above, it is difficult to model the GHG emissions implications of increasing biofuel demand, and harder to find agreement about the hierarchy of impacts between different feedstocks. Predicting feedstock-level impacts on food security or biodiversity is no easier and has not received the same level of analytical investment. Nevertheless, it would be possible in principle to consider the development of some form of aggregate metric to express not only the risk of ILUC emissions from different biofuels, but also the level of impact of food markets and/or biodiversity.

Food security impacts could be analysed building on the same economic modelling tools that are used to produce land use change scenarios. The economic models operate through assumed price changes, and it is possible to analyse the expected impact of the modelled price changes on metrics such as poverty rates. For example, Wiggins et al. (2008) present estimates of the potential increase in poverty rates that could be caused by food-commodity price increases due to biofuel demand identified in outputs from economic models. If the details of regional price changes for each feedstock scenario in outputs from the main ILUC modelling exercises for the European Union could be obtained from the modelling teams (Laborde, 2011; Valin et al., 2015), then it would be possible to use food security and poverty modelling tools to assess whether there is any significant difference expected between the different feedstocks. While there is an extensive literature available associated with the food price and food security impacts associated with biofuels in general, as far as we are aware there is limited literature directly comparing the impacts of different feedstock choices, and therefore developing results that could be used as the basis for a support metric would be expected to require funding original research.

Similarly, it ought to be possible to draw conclusions about potential biodiversity impacts by cross-referencing the regions in which crop expansion and intensification are predicted in ILUC modelling exercises with information about where the threat to biodiversity from agriculture is lesser or greater.

Having constructed impact metrics for food security and biodiversity impact to go alongside estimates of ILUC emissions, an aggregate metric could be constructed by applying some weighting and normalisation to the scores, resulting in an aggregate impact score for each category. The construction of such aggregate impact metrics is not unusual in lifecycle analysis, for example Zah et al. (2007) presents aggregated environmental impact metrics comparing biofuels to fossil fuels, showing that when additional environmental impacts are included alongside GHG emissions biofuel production chains may not be environmentally preferable to fossil fuels.

Having constructed this aggregate metric, support could be varied using policy levers such as awarding credit in proportion to the achieved score, setting sub-targets for fuels beating some threshold score or restricting the contribution of fuels with lower scores.

8.6.1 Compatibility with the RED

While there is an obvious appeal in seeking to construct a broader metric for assessing the performance of biofuels, such approaches are not foreseen in the Renewable Energy Directive and would go beyond the scope of Article 26(1). The European Commission would be likely to consider Member State initiatives to integrate broader environmental and social concerns into the systems for determining levels of support for different biofuel feedstocks as going beyond what is permitted within the RED II.

8.6.2 Administrative burden

Developing compound environmental/social impact metrics would be a major analytical and consultative exercise for the Danish Government, potentially requiring comparable resourcing to that which would be needed to undertake original ILUC modelling. Such metrics could be expected to be subject to persistent stakeholder engagement and challenge, and would need to be updated periodically to reflect changing situations. Introducing such a system would require a long-term commitment. As with GHG targets in a hybrid LCA implementation, work would be required to establish what level of targets would be necessary to meet EU level targets when using a 'compound metric' system for rating fuels. In terms of day-to-day administration, the burden would depend on the form of implementation. If implemented analogously to an energy-based mandate the administrative burden would not be fundamentally greater, but the added complexity would likely require the availability of additional staff, at least one FTE (full time equivalent).

8.7 ILUC and the Product Environmental Footprint framework

The Product Environmental Footprint (PEF) has been developed by the European Union as part of the 'Single Market for Green Products Initiative'. The intention of the PEF framework is to provide a single standard for environmental performance claims to provide confidence to consumers and reduce costs for businesses.

The PEF framework relies on the use of lifecycle analysis tools but presumes the use of attributional rather than consequential approaches. The PEF framework identifies ILUC emissions as generally out of scope: "Greenhouse gas emissions that occur as a result of indirect land use change shall not be considered unless PEFCRs explicitly require to do so. In that case, indirect land use change shall be reported separately as Additional Environmental Information, but it shall not be included in the calculation of the greenhouse gas impact category" (European Commission, 2013). As we understand the PEF framework, it does not move the ILUC debate forward – rather, it provides another instance where we are challenged to find a way to integrate ILUC information from consequential assessment within a primarily attributional framework. We would anticipate that, to the extent that it deals with biofuels, the PEF framework is likely to follow the lead of the RED, rather than create a lead to follow.

8.8 Review of the pros and cons of the options

Table 6 Summary of the pros and cons of the identified regulatory options to distinguish biofuels based on ILUC emissions

Option	Pros	Cons	Acceptability to EU
Capping food-oil-based fuels	Simple Directly reduce expected ILUC emissions	Blunt instrument Ignores expected ILUC from ethanol crops	This approach is explicitly allowed in the RED II
GHG-based support with a hybrid LCA	Provides a more nuanced adjustment of support Consistent with a technology neutral approach Make reported GHG savings more consistent with a best estimate of delivered GHG savings	Requires choosing and justifying a single set of ILUC factors Adds greater complexity to the regulation Would make Danish domestic GHG accounting inconsistent with the rest of the EU	The Commission is likely to point to Recital 81 as a basis to argue that this regulatory approach goes against the intentions of the EU institutions
Hybrid LCA in assessing compliance with sustainability criteria	Would effectively exclude food-oil based fuels Would force improvements in GHG performance for some ethanol pathways	Has an effect similar to simply capping food-oil based fuels By limiting eligibility of both biodiesel and ethanol could make it more difficult to deliver targets	The RED II explicitly defines the GHG accounting rules for assessing compliance with minimum GHG thresholds, adding ILUC emissions would be inconsistent with those rules.
Adding an 'intermediate ILUC-risk" category	Build on an established analytical approach that avoids direct reliance on ILUC modelling. Categorisations could be updated if Commission analysis evolves.	Currently, would only affect soy oil unless threshold was set very low It is not clear that lower ILUC risk scores (around 2%) are well correlated with expected ILUC impacts	Likely to be acceptable if clearly justified. Setting a very low threshold value might be seen as a concern by the Commission if it led to restrictions on fuels with lower modelled ILUC emissions.
Support for certified low ILUC-risk fuels	Certification provides a basis to believe that ILUC emissions are avoided or much reduced. Encouraging adoption of low ILUC-risk measures could encourage wider adoption of better agricultural practices.	Low ILUC-risk systems are still being developed - certification schemes are not yet finalised and supply chains do not yet exist. A poorly implemented low ILUC-risk scheme may fail to truly reduce ILUC and could become a loophole.	Likely to be acceptable
Multi-criteria socio-environmental performance scores	Allow regulations to reflect a broader characterisation of the environmental and social performance of fuels. If well implemented, could drive optimal outcomes.	Complex, and any system for compiling performance scores would inevitably be challenged as subjective. Stakeholders may object to the expansion of policy goals beyond climate change mitigation. Resource intensive to develop.	As such a system would use more information than just ILUC emissions estimates this would go beyond the scope offered by Article 26(1).

9 Outline for a Danish regulatory approach to reduce ILUC impacts

Above, we discussed a number of options to address ILUC in biofuel regulation. In this section we present the outline for a system based on combining three of these elements:

1. Eliminating incentives for biofuels expected to cause the highest ILUC emissions (palm- and soy-based fuels);
2. Limiting the contribution from other food-oil based fuels;
3. Adding additional support for certified low ILUC-risk fuels through exemption from this limit.

We believe that this approach could significantly reduce ILUC emissions from Danish biofuel use, would be relatively simple to implement and would be consistent with the legal text of the RED II and acceptable to the European Commission. The first element of this portfolio, removing incentives for the use of palm- and soy-based fuels, has already been confirmed by the Danish Government. This decision is supported both by the balance of the best available evidence from ILUC modelling and by the results of the high ILUC-risk assessment.

The second element, limiting the contribution to targets from food oils other than palm and soy, would primarily affect the use of rapeseed and sunflower oil for biofuel production. Danish energy statistics for 2019 show that biodiesel constituted about 3.5% of total transport energy supply (including aviation and maritime). A cap could be introduced on the contribution of biotickets for food-oil-based biodiesel reducing to, for example, 1.5% by 2030. In a GHG based system, this could be converted to a cap of 1% on the contribution to delivered GHG reductions. The exact level for the cap should be based on a consideration of the development of the Danish transport energy supply.

The third element would be to exempt certified low ILUC-risk feedstocks from this limit. This would mean that low ILUC-risk rapeseed, sunflower soy and/or palm oil would have a clear market opportunity in Denmark, competing for space in the biodiesel market with waste and residual oils. In contrast, this would create no incentive for low ILUC-risk ethanol feedstocks as food and feed crops would still be eligible to be used up to the food cap.

This combination of measures would create a clear trajectory for the Danish biofuel policy. It would be expected to create a two-tier market in GHG-reduction biotickets, and this would go some way to resolve some of the technology neutrality problems that arise from not including ILUC emissions in the LCA. First generation biofuels would still get over-credited on a GHG basis compared to fuels with no expected ILUC emissions, but the quantities of fuels benefitting from this advantage would be constrained by the ethanol blend wall and the imposed limit on food-based biodiesel. While there may still be indirect emissions associated with the other categories of renewable transportation fuels competing for the remainder of the market space, these can be expected to be smaller.

10 Discussion

Issues of the carbon cost of land use are an unavoidable part of analysing biofuel policy. For the last twelve years the European Union has been trying to navigate a path between biofuel optimists and advocates who see ILUC as a dangerous distraction delaying the deployment of an important tool in the fight against climate change, and biofuel pessimists and critics who are concerned that largescale bioenergy is a climate non-solution that will fail to deliver significant net GHG reductions while destroying habitats and undermining food security. ILUC emissions analysis has been deployed in the effort to identify whether food-based biofuels can really be a useful climate solution or whether they should be curtailed.

As the RED II comes into operation, the broad policy conclusion of the European institutions is that food-based fuels are not going to be a major long-term contributor to European energy supply or climate goals, but simultaneously that they can have a supporting role to play in reducing the GHG intensity of liquid transport fuels and need not have their subsidies completely removed. It is clearly understood that we expect some biofuels to deliver more benefit than others. The EU has concluded that ILUC analysis does not and cannot furnish us with precise indisputable answers, but equally that ILUC is a problem that should not and cannot be ignored.

The weight of evidence compiled for the European Commission over this period points to some conclusions that can be made with a moderate degree of confidence:

1. Assessment of ILUC emissions should inform the levels of support given to biofuels by policy makers;
2. Food and feed crops are associated with a greater risk of ILUC emissions than energy crops;
3. Vegetable oils are associated with higher estimated ILUC emissions than starchy or sugary feedstocks;
4. Palm oil carries the highest risk of large ILUC emissions and probably has higher ILUC emissions than other vegetable oils;
5. Soy oil carries the second highest risk of large ILUC emissions and probably has ILUC emissions between those of palm oil and those of other vegetable oils.

EU biofuel policy has already been revised since 2009 to address the risk of ILUC. This has involved scaling back ambition for the use of food-based biofuels, developing new incentives for the development of advanced biofuel technologies, and recently adopting the high ILUC-risk assessment, based on which support for palm oil biofuels is due to be limited.

Within the guidelines set by the RED II, there is a considerable degree of discretion for European countries to steer their biofuel industries more or less firmly away from food-based fuels, and to take more or less account estimated ILUC impacts when implementing the RED II. This can be done by setting lower limits on total national consumption of food-based fuels. It can be by giving more support to advanced biofuels or to other ways of delivering renewable energy to transport (including electric vehicles and electrofuels). And it can be done by exercising the discretion given by Article 26(1) to 'distinguish' between food-based fuels based on ILUC emissions.

In this report we have discussed the evidence upon which a Member State may base measures to distinguish between feedstocks, and we have discussed a number of regulatory approaches that could be used to use that evidence to intervene in the biofuel market. The simpler approaches would involve imposing limits on the supply of fuels identified as being likely to be associated with higher ILUC – this could be achieved by setting limits on support for vegetable oil fuels, by including estimated ILUC values in assessment of compliance with minimum GHG saving thresholds, or by creating a new category of 'intermediate ILUC-risk' fuels based on the Commission's high ILUC risk assessment. We have also discussed the possibility of using a hybrid LCA approach to determine levels of support for biofuels from each feedstock, echoing the regulatory approach that has been used with some success in California. We have provided an outline for a regulatory approach that would combine restrictions on total use of food-oil based fuels with an increased role for low ILUC-risk certification.

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