

A Guide for the Perplexed to the Indirect Effects of Biofuels Production

Chris Malins, Stephanie Searle, and Anil Baral




THE INTERNATIONAL COUNCIL ON CLEAN TRANSPORTATION

www.theicct.org

communications@theicct.org

BEIJING | BERLIN | BRUSSELS | SAN FRANCISCO | WASHINGTON

ACKNOWLEDGMENTS

The authors acknowledge the invaluable contributions of the following people and organizations in the completion of this project. While this report would not have been possible without the engagement of these individuals and institutions, they do not necessarily endorse any or all of the conclusions expressed herein, which represent the position of the International Council on Clean Transportation only.

Reviewers: David Laborde, Ignacio Vazquez.

Other contributors: Peter Witzke, Chris Sichko, Gaurav Bansal, Drew Kodjak, John German, Richard Plevin, Ralph Heimlich, Nic Lutsey, Jacinto Fabiosa, Steve Berry, Wolfram Schlenker, Robert Edwards.

Funder: The ClimateWorks Foundation.

Cover photographs via Wikimedia Commons:
Oil Palm Plantation in Cigudeg-03, by Achmad Rabin Taim, Jakarta, Indonesia (P3260481)
Rapeseed, Champ de colza, Côte-d'Or Bourgogne, Avril 2014, by Myrabella
Soybean Plantação de soja, by Coloradogoias
Minnesota Corn (20030826), by Kris from Seattle, USA (Minnesota)
In the Corn Field, by Lars Plougmann (originally posted to Flickr as In the corn field)

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

EXECUTIVE SUMMARY

ES.I. BIOFUELS POLICY OVERVIEW

Since 2000, the global biofuel supply has leapt from less than 20 billion to more than 100 billion liters a year (Figure A). Biofuels now make up about 3 percent of global transportation energy use. This expansion has not been market driven—in general, biofuels remain more expensive than their fossil fuel alternatives—but led by government interventions (Table A), in particular, the Renewable Fuel Standard (RFS) in the United States and the Biofuel and Renewable Energy Directives in the European Union (EU). These policies have been put in place with three objectives in mind: reducing overall carbon dioxide emissions, improving energy security by diversifying supply, and supporting rural incomes.

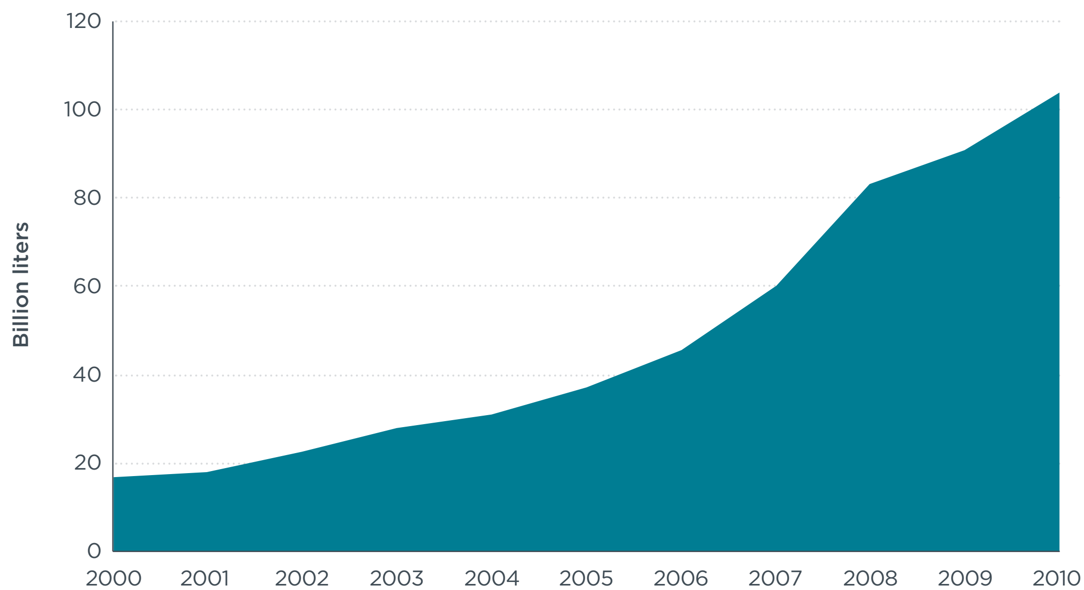

Figure A. Global biofuel supply, 2000–2010

Source: International Energy Agency, 2010

While government policy has been successful in encouraging the widespread adoption of biofuels, it is less clear whether biofuels policies are fully delivering on the primary objectives. In particular, there has been extensive discussion in recent years about whether biofuels policies have delivered net greenhouse gas emissions reductions. U.S. and EU biofuel mandates may not currently be reducing carbon emissions. Scientific discussion is ongoing about certain questions in direct life cycle analysis, such as how to quantify the climate impact of nitrous oxide emissions from fertilized agriculture. Perhaps the most controversial debate, however, has been on the land use implications of biofuel mandates and the associated release of carbon into the atmosphere. Among the policies to promote biofuels shown in Table A, only those of the United States, California (which has been given license by the federal government to set its own rules), and the EU have established carbon intensity requirements and sought to characterize the indirect land use changes of those fuels.

Table A. Global biofuel support policies in selected regions

Jurisdiction	Policy name	Biofuel-related targets
European Union	Renewable Energy Directive (RED)	10 percent renewable energy in specified transport modes by 2020
	Fuel Quality Directive (FQD)	6 percent reduction in fuel GHG emissions intensity by 2020
United States	Renewable Fuel Standard (RFS) 2	136 gigaliters (GL) per year of biofuels by 2022
California	Low Carbon Fuel Standard (LCFS)	10 percent reduction in fuel GHG emissions intensity by 2020
China	National plan	Ethanol: 3.8 GL/yr by 2010, 12.6 GL/yr by 2020; Biodiesel: 337 megaliters (ML)/yr by 2010, 2.3 GL/yr by 2020. 10 percent ethanol in 10 provinces
India	National Policy on Biofuels	10 percent biofuel blending by 2008, rising to 20 percent by 2017
Canada	Renewable Fuel Standard	5 percent ethanol in gasoline by 2010; 2 percent biodiesel in diesel and heating distillate oil by 2012
Mexico	Law for the Promotion and Development of Bioenergetics	2 percent biofuels in Guadalajara by 2011 and in Monterrey and Mexico City by 2012
Japan	Biomass Nippon Strategy	0.5 GL/yr by 2010; nominal targets of 2 GL/yr by 2020, 4 GL/yr by 2030
Australia	Energy Grants (Cleaner Fuels) Scheme; Ethanol Production Grants	350 ML/yr of biofuels by 2010
Brazil	Mandatory biodiesel requirement; Ethanol fuel program	5 percent biodiesel blend by 2010; 25 percent ethanol by 2007

ES.II. INDIRECT LAND USE CHANGE, AND INDIRECT LAND USE CHANGE MODELING

Often, people refer to biofuels as ‘carbon neutral’ even though the carbon emissions from vehicle tailpipes when using biofuels are the same as from fossil fuels. This notion of carbon neutrality is based on the idea that the carbon released in combustion when biofuels are used was sequestered recently from the atmosphere rather than freed up from stores of fossil carbon. However, this would be correct in practice only if the process of biofuels production did not itself cause any release of stored carbon. In reality, much of the biofuel supply chain is dependent on fossil fuels, and the process of cultivating biofuels requires land—land with carbon stored in soils and in biomass, which will sequester carbon naturally if left uncultivated. For biofuels to work as a climate change policy, the sum of the releases of stored carbon caused by biofuels production at every stage (from soils and biomass on cultivated land, from fuel burned by tractors and processing plants, and so forth) and any associated emissions of non-CO_2 greenhouse gases (such as nitrous oxide released as a result of nitrogen fertilizer application) must be less than the carbon emitted by using fossil fuel gasoline and diesel.

Assessing the land use change emissions from biofuel policies is difficult because the crops used for biofuel feedstock do not necessarily come from the areas where land use change occurs. Biofuel mandates require either an increase in agricultural production or a reduction in consumption of feedstocks by other sectors (a simple mass balance problem). If feedstock is made available because use of crops for food is reduced, there is no land use change problem but there is a conflict with food security. If, on the other hand, absolute levels of feedstock production rise, this will generally require an increase in land that is put to agricultural use. It may be that production from existing fields is used for biofuels and that new land is converted to fill the subsequent deficit in food supply—but, across the system as a whole, biofuels policy will have caused land use change. Because these changes can happen at some distance from the location of biofuel feedstock cultivation, they are referred to as indirect land use change (ILUC).

In the literature, it is common to draw a distinction between ‘direct’ and ‘indirect’ land use changes. A direct land use change is one that relates to an area where biofuel feedstock is physically grown. For instance, if a pasture area in the United States is replaced by corn, and that corn is used for ethanol production, that is a ‘direct’ land use change. Identifying a land use change as direct does not tell you about causation, that is it does not require that the reason for making the decision to farm additional area was increased corn demand for ethanol. When we talk about indirect land use change, on the other hand, the objective is to compare some ‘baseline’ of a world with less biofuel demand to a ‘scenario’ of a world with more biofuel demand, and consider how land uses have changed (or might change) across the whole system as a result. In the real world, researchers can use econometric analysis to investigate evidence for a causal link between demand and land use change. In models, the causation is built in by hypothesis—if the independent variable that changes from the baseline to the scenario in a model is biofuel demand, then resulting land use changes in the model must have been caused by the demand change.

Indirect land use change is complicated. The supply chains for some agricultural commodities stretch halfway across the world; modern agricultural markets are linked so that there cannot be an increase in corn prices only in the United States; those price rises will be felt in Mexico, in South America, even in Asia, Europe, and Africa. As prices change, consumption shifts to other grains (wheat, rice, and so on), changing their prices as well. The reality of ILUC in response to increased biofuel demand is thousands of commercial decisions being taken all through the supply chain and across the global market, driven by the interaction of prices, government policy, regulations, trade relationships, and market expectations. Added together, these decisions result in changes to agricultural commodity prices, changes in how much people eat and what they eat, changes to how much energy people use—and they ultimately expand the agricultural frontier. Modeling ILUC is a process of outlining a scenario for how this complicated global system might respond to an increase in demand for biofuels.

ES.II.i Why ILUC is important for policy

A central objective of biofuel policy is to reduce carbon emissions from the transportation sector. It is therefore important to conduct life cycle analysis (LCA)[1] of biofuel pathways in order to compare their carbon intensity to that of the fossil fuels they will replace. The traditional approach to the LCA of biofuels has been to consider all of the steps, from sowing a biofuel crop to delivering the fuel to a vehicle, to assess the inputs of energy and materials at each stage, and to sum these to give the life cycle carbon intensity. Traditional LCA considers most fundamental inputs to biofuels production—nitrogen fertilizer, pesticides, tractor fuel, process fuel, etc.—but it takes something else that is equally fundamental for granted. Traditional LCA has always implicitly ignored the carbon implications of using the land itself, the ILUC. The problem is that land is a fundamental requirement for producing biofuels, but land use is not normally carbon neutral—natural landscapes contain large amounts of stored carbon not only in plant matter above ground but often even more below ground, in soils. Moreover, these tracts frequently have the natural capacity to sequester still more carbon over time.

By its nature as a market-mediated, indirect effect, ILUC cannot be measured precisely or directly. It is possible to analyze historical data for evidence that ILUC has occurred and to get a sense of the order of magnitude of emissions, but to quantify ILUC, modeling approaches are necessary. If it were clear that CO_2 emissions from ILUC were greater than any possible carbon savings from using biofuels, the conclusion would be simple—there would be no case for biofuel policies on a climate change mitigation basis. Alternatively, if it were clear that ILUC emissions were negligible compared with the potential savings from displacing fossil fuels, the conclusion would also be simple—policy would only need to consider direct emissions. The reason that the ILUC discussion remains so important to policymakers is that the modeling work carried out to date puts ILUC emissions on the same scale as the potential savings. As a result, some biofuel pathways deliver significant climate benefits while others do not, and ILUC will often be what makes the difference. Figure B shows that analyses performed by or for regulators in the United States and Europe agree that there will be significant ILUC emissions for all food crop pathways. However, while some biofuels are still expected to offer significant emissions savings, others (notably several biodiesel pathways) offer negligible savings or may even be worse for the climate than the fossil fuels they replace.

1 A biofuel life cycle analysis adds up the greenhouse gas emissions associated with all stages of the biofuel production process that lie within the defined "system boundary." Setting a system boundary (defining what is included and excluded) gives clarity to the analysis but in some cases may result in significant emissions sources, such as land use changes, being overlooked.

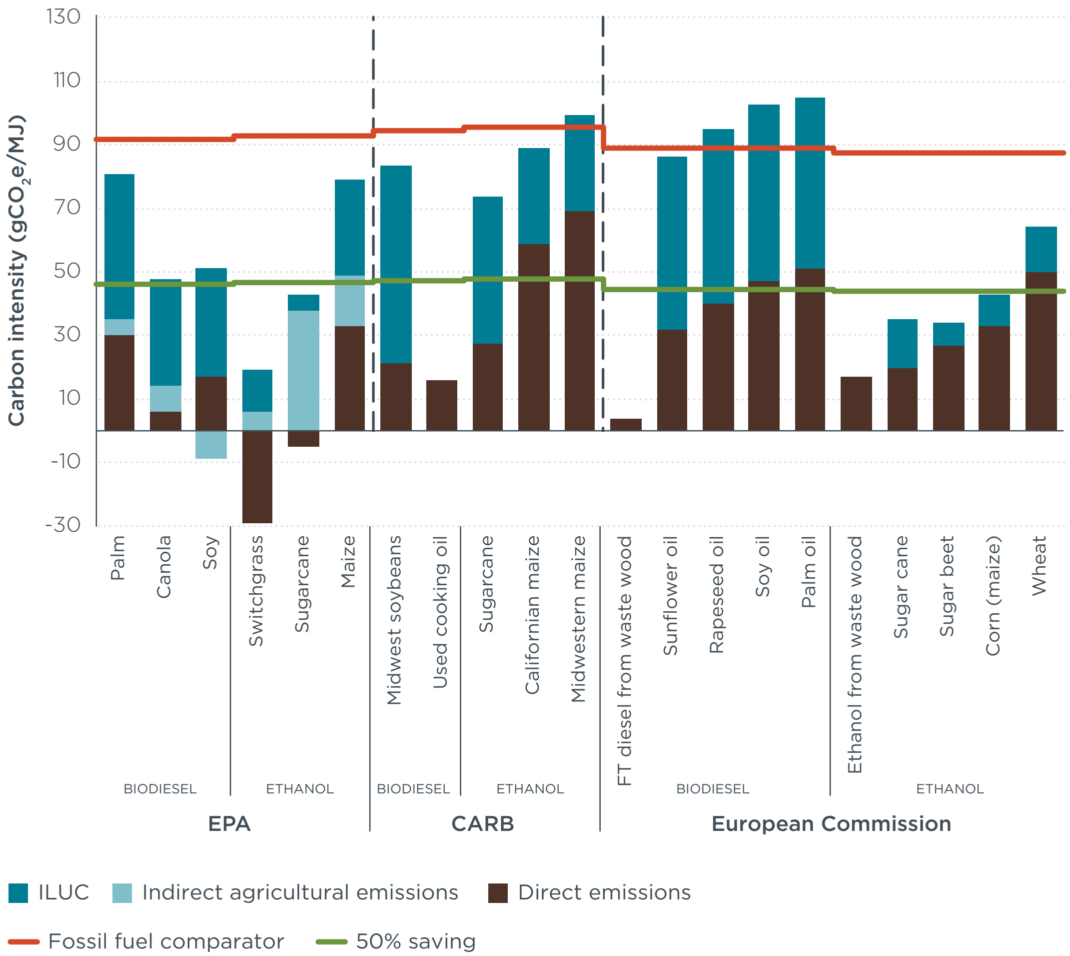

Figure B. Total emissions of modeled biofuel pathways, compared against fossil fuel carbon intensity

ES.II.ii ILUC models, and why you may be perplexed

An ILUC model is any model that attempts to analyze the consequences of increased demand for biofuels on land use in the economy as a whole. Some authors have suggested relatively simple, spreadsheet-based models of ILUC—for instance, the ‘risk adder’ approach proposed by the Öko-Institut (Fritsche 2010), based on historical extrapolation, or the ‘causal descriptive’ model of E4tech (E4tech 2010), based on expert opinion. Others have used more complex economic models. For example, ‘partial equilibrium’ models include the whole agricultural sector for a country, set of countries, or even the entire world. General equilibrium models go further, encompassing not only agriculture but all major sectors of the economy. In many cases, economic models originally developed to analyze trade agreements and the implications of agricultural policy change have been developed and refined to address questions about biofuels and carbon emissions. In economic modeling exercises such as those found in Laborde (2011a), Hertel et al. (2010a), and U.S. EPA (2010a), a scenario involving higher biofuel demand is compared with a baseline (counterfactual) with lower biofuel demand. Any difference in land use between the two scenarios can then be attributed to biofuels as ILUC. In general, the economic models do not have assumptions about the carbon implications of land use change built into them, so the outputs in terms of the area undergoing change must be combined with an estimate of land-based carbon stocks to calculate emissions values.

The choice of model defines the possible responses to increased agricultural demand, and this in turn defines the set of carbon sequestration and emissions impacts that are analyzed. For example, in equilibrium economic modeling, one key response to increased demand for feedstock from the biofuel sector is reduced demand from the food sector—that is, there is competition between food and fuel. On the other hand, E4tech (2010) chose not to consider possible reductions in food consumption in its causal descriptive spreadsheet model. This choice places one form of carbon saving off-limits. There can be more subtle differences, too—for instance, one model might use a more complex set of land categories than another, or one might consider forgone carbon sequestration and another might not, or one might model emissions from peat drainage and another might not. Each model is built on tens, hundreds, or even thousands of explicit or implicit decisions, assumptions, and input parameters, and even for well-documented models it can take considerable effort for other experts, let alone nonspecialists, to understand what is driving the results.

Given the wide variation in modeling approaches, it is unsurprising that there is a correspondingly wide variation in model outcomes. Figure C shows point estimates for the magnitude of ILUC emissions reported in a range of studies from the literature. As shown, there are many estimates, for many biofuel pathways, by many models. While the bulk of emissions intensity estimates fall in a range from about 10 to 80 grams of carbon dioxide equivalent per megajoule (gCO_2e/MJ), a few studies have arrived at extremely high values (higher for instance than the estimate of 104 gCO_2e/MJ for corn ethanol originally reported by Searchinger et al. [2008]). On the other hand, a few have suggested that ILUC emissions could be negative—that is, for certain pathways land use change could result in additional carbon sequestration. There is a tendency for biodiesel estimates to be higher than ethanol estimates, and this is reflected in the range of ILUC emissions intensity values proposed by regulatory agencies, which is also shown in Figure C. For context, the full lifecycle emissions intensity of fossil gasoline and diesel is normally around 90 gCO_2e/MJ.

There is not enough space here to consider and evaluate every study of ILUC that has been done. Rather, it is the aim of this report to provide a basis to understand and evaluate claims about indirect land use change, in order to help policymakers, regulators, and others weigh the value of the evidence in front of them.

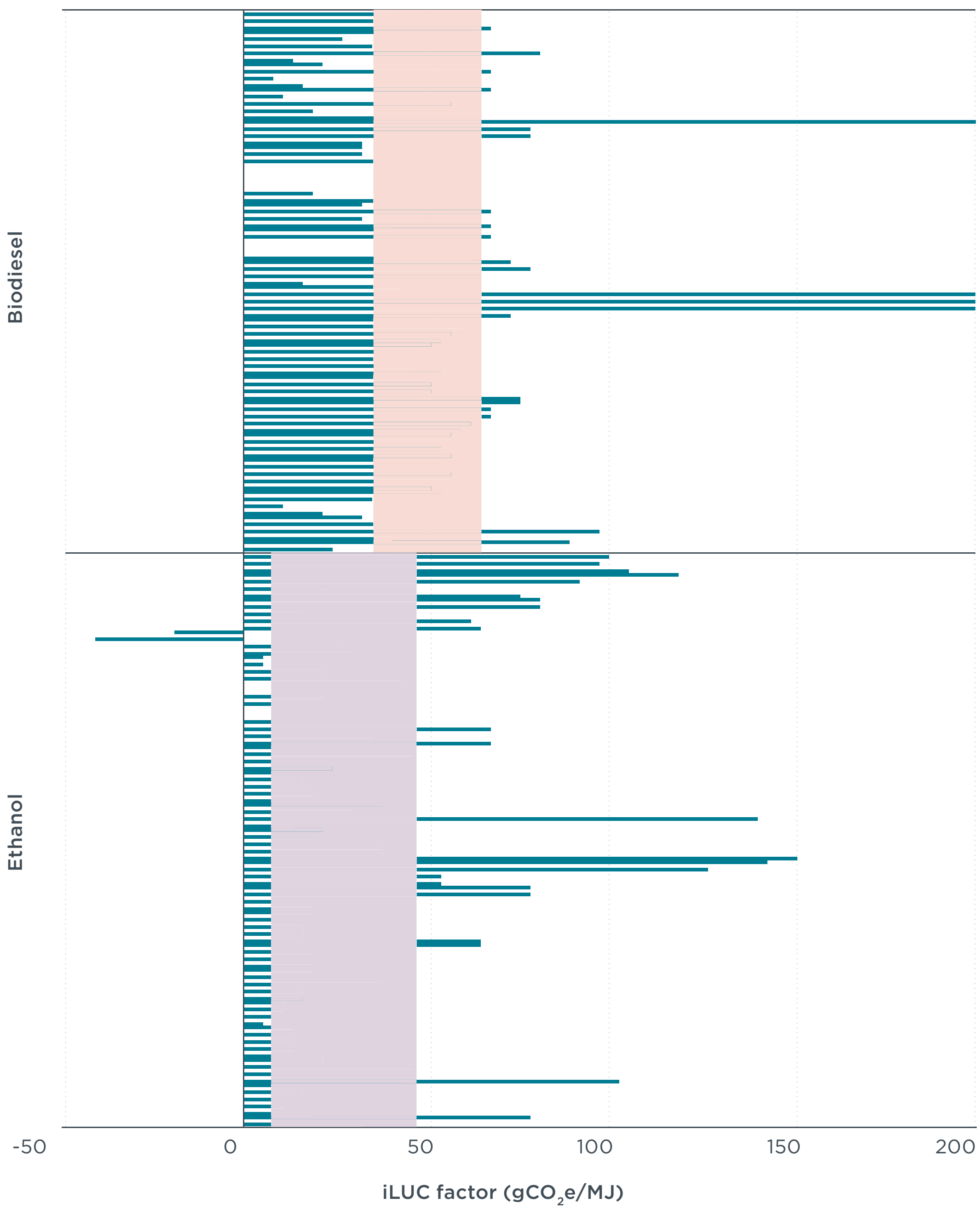

Figure C. Variation in ILUC emissions intensity from different modeling exercises

Notes: The ranges shown for regulatory estimates of ILUC emissions intensity are based on the California LCFS and U.S. RFS rules and the Laborde (2011a) analysis that is the basis for proposed ILUC factors in Europe.

Some estimates go even higher than 175 gCO_2e/MJ, but this graphic does not show the full magnitude of those very high estimates. This report aims to include as comprehensive a characterization of the range in the literature as possible, but inclusion does not imply that the International Council on Clean Transportation considers any particular value to be credible.

Sources: Results taken from Edwards, Mulligan, and Marelli (2010), E4tech (2010), Akhurst, Kalas, and Woods (2011), Searchinger et al. (2008), U.S. EPA (2009; 2010a,b,c, 2012), Fritsche (2010), ARB (2009), Tyner et al. (2010), Hertel et al. (2010a), Taheripour, Tyner, and Wang (2011), Al-Riffai, Dimaranan, and Laborde (2010a), Laborde (2011a), Tipper et al. (2009), Dumortier et al. (2009), Britz and Hertel (2011), Wang et al. (2011; 2012), Dunn et al. (2013), Overmars et al. (2011).

ES.III. A GUIDE FOR THE PERPLEXED—WHY ARE THE ANSWERS DIFFERENT?

> The generosity of the Earth allows us to feed all mankind; we know enough about ecology to keep the Earth a healthy place; there is enough room on the Earth, and there are enough materials, so that everybody can have adequate shelter; we are quite competent enough to produce sufficient supplies of necessities so that no one need live in misery.
>
> —E. F. Schumacher, from A Guide for the Perplexed (Harper Publishing, 1977)

> We naturally like what we have been accustomed to, and are attracted towards it.... The same is the case with those opinions of man to which he has been accustomed from his youth; he likes them, defends them, and shuns the opposite views.
>
> —Moses Maimonides, from The Guide for the Perplexed (12th century A.D.)

> It turns out that despite the complexity of the models, only a handful of easily understandable assumptions are important in determining the simulation results. By showing the effect of these assumptions on the predicted economic costs, not just in one particular model but in all of them, this report can help readers to apply their own judgments about which models are more realistic and to reach their own conclusions about which economic predictions are more credible.
>
> —Robert Repetto and Duncan Austin, from The Costs of Climate Protection: A Guide for the Perplexed (World Resources Institute, June 1997)

The computational general and partial equilibrium economic models that have been used to estimate ILUC emissions by regulators in the United States and the European Union are built on the back of decades of research into agricultural and economic systems, composed of myriad mathematical relations based on economic theories that continue to be discussed, debated, and refined. These models provide a sophisticated but still simplified view of market interactions—in some ways an idealized view. Nevertheless, a rigorous, well-vetted model provides insight into how markets are likely to behave, and it can produce quantitative estimates of the emissions (and price and land use, etc.) that policymakers should expect from biofuel mandates. The specifics of these models are important, but they can become a barrier to comprehension rather than an aid to understanding if the level of detail becomes overwhelming. Without explanation, an ILUC model can seem to be a mysterious black box; on the other hand, having a thousand input parameters listed explicitly might not bring one any closer to understanding the meaning of the modeling results, either.

This study is not the first, and doubtless will not be the last, to review the subject of indirect land use change. The British government's Gallagher Review (UK RFA 2008) assessed six questions about the likely impacts of biofuel demand, with the aim of drawing conclusions about the nature and level of the risks and making recommendations on how these risk should be dealt with in government policy and carbon accounting. The European Commission's Joint Research Centre (Edwards, Mulligan, and Marelli 2010) approached the question by setting up a framework to compare the results of various economic modeling exercises. The European Commission Directorate-General for Energy (EC DG Energy 2010) produced a literature review focused on comparing "methodological and data choices" in various studies attempting to quantify indirect land use change. The U.S. Department of Agriculture reported to Congress (Marshall et al. 2011) on the state of knowledge about the drivers of land use change and the models used to assess it, with the intention of providing a neutral survey of the literature. Ecofys, a renewable energy consultancy, reported to the Global Bioenergy Partnership (Ecofys 2011) as part of its discussion of the indirect effects of biofuels production, with a focus on explaining modeling attempts to quantify indirect effects and exploring options to avoid or minimize them. California's Air Resources Board Expert Workgroup on Indirect Land Use Change (2010) produced a series of reports considering in detail a range of questions critical for California's modeling of ILUC. There are other papers and reports available covering the same areas to a greater or lesser extent.

This report does not aim to be a comprehensive literature review in the style of the DG Energy or U.S. Department of Agriculture reports. It is not built around a new quantitative analysis in the way that the JRC model comparison is. Rather, as the World Resources Institute (Repetto and Austin 1997) did for the cost of climate action in *The Costs of Climate Protection: A Guide for the Perplexed,* this report aims to identify and explore the key factors that determine the amount of indirect land use change occurring in the real world. Based on this analysis, it looks at how these factors are represented in the major models being used for regulatory purposes and how they influence the estimates on which policymakers must base their decisions. The scope of the report has been limited in order to focus on the determinants of how much land use change will occur. Broader carbon accounting questions, such as time accounting and assessment of other indirect effects, are beyond that scope, as is a comparative evaluation of the policy responses available to deal with ILUC.

ES.III.i. The key factors

While the agricultural economic models are extremely complicated, the basic questions they are used to answer can be quite simple—when demand for agricultural products increases, is more land brought into production, and if so how much land and what are the carbon consequences?

It is an axiom of economics that an increase in demand for a product from one sector must be met by some combination of an increase in supply and a reduction of demand for the product from other sectors. In the case of an increase in demand for food commodities, raising supply means either improving the productivity of existing agriculture or bringing new land into production. A reduction in demand outside the biofuel sector generally means people eating less.[2] The amount of ILUC that occurs depends on the balance between productivity increase, land expansion, and food demand change. The analysis here identifies five factors that determine this balance and a sixth that governs the carbon emissions resulting from the land use change.

The first factor, determining whether people eat less food as prices change, is ***elasticity[3] of food demand to price.*** If food consumption is highly elastic relative to price, it means that a small price increase spurred by biofuel demand would cause people to eat much less, making the uneaten material available for conversion to biofuel. On the contrary, if food demand is relatively inelastic to price, then even a large price change would have

2 This reduction in food consumption can be direct (e.g., people eat less bread and hence require less wheat for flour) or indirect (e.g., people eat less beef, therefore livestock producers need less wheat for cattle feed). There can also be a reduction in other uses for crops, such as brewing beer or making soap, but food is by far the largest part of the equation, and it's the focus here.

3 In economics, the elasticity of some quantity (such as demand or yield) to price is a measure of how much that quantity changes if prices change.

little effect on diets, and the material for biofuels production would need to come from somewhere on the supply side.

Working out how the productivity of the agricultural system changes as a result of new crop demand is more complicated. Three parameters interact to determine the overall productivity effect (and each of these could in theory be broken down even further). The first of these is the ***elasticity of yield to price***. This factor determines how the average yield of a given crop changes in response to a price change. If raising yields is easy, and farmers are very responsive to price, yield increase can deliver the raw material to make biofuels without requiring large areas of new land or forcing people to eat less. If on the other hand raising yields is difficult, or farmers are unresponsive to prices, then each field will still produce the same amount of crops as before, and the biofuel feedstock will need to come from new land or reduced consumption.

The second determinant of productivity is ***choice of crops.***[4] Some crops are more productive, in terms of metric tons of raw material produced per hectare, than others. The nutritional content may vary, but if the focus is on sheer quantity of production then growing more of the highest-yielding crops, and less of crops with low yields, can change the productivity of the system as a whole.

The third factor is ***utilization of co-products.***[5] Consider corn, from which only about two-thirds of the material in the grain can be fermented into ethanol. The rest, including proteins and fats, is left over for other uses or disposal. The least productive fate for this co-product would be simple discarding; an alternative use might be to burn it for heat and power, but the standard current usage is as animal feed. This use is also the most efficient in terms of limiting ILUC, as it reduces demand for other grains. If co-products are well utilized, that is equivalent to increasing the productivity of the land.

If productivity gains are not adequate to meet the need for increased feedstock supply, the rest must come from growing crops on new land. The rate at which land expansion occurs is determined by the ***elasticity of area to price.*** If area is greatly price elastic, it means that farmers will respond to a small price change by clearing and cultivating lots of new land, so that there will be little need to increase yields or eat less, and the ILUC will be large. In contrast, if area elasticity is small compared to food consumption and yield elasticity, then only a small area will be converted.

The final factor affecting the magnitude of ILUC emissions, is the ***carbon stock of new land.*** The balance of productivity increase, demand reduction, and land expansion dictates how many new hectares are needed, but translating that into an ILUC emissions intensity requires knowing (or estimating) how much carbon is released when that land area is cleared. If new land is taken primarily by clearing forest, the emissions are likely to be high. If new land can be found with sparse vegetation and low levels of organic carbon in the soil, the emissions will be much less.

The importance of these six factors in determining the amount of land use change that happens can be seen by qualitatively comparing two starkly different scenarios of how the agricultural system might respond to biofuel demand. The first to be considered is

4 This can include choosing whether to raise cattle on pasture, which has a lower productivity in terms of food yield per hectare, or to grow crops for direct human consumption, which is much more productive.

5 Co-products are different products derived from the same feedstock. For instance, in the case of oilseeds, one co-product from oil pressing is vegetable oil (which can be used for biodiesel), while a second is oil meal (which can be fed to livestock).

from the original ILUC modeling exercise by Timothy Searchinger et al. (2008),[6] which claims that corn ethanol delivers no carbon benefit. The second is from Bruce E. Dale et al. (2010), in which, rather than modeling ILUC, the authors outline a vision of how biofuels production could be structured to deliver significant benefits without ILUC emissions. Table B lays out the ways in which the two papers treat the six factors differently. Ultimately, Searchinger's team conclude that ILUC renders biofuel policies ineffective, while Dale and his fellow authors find that ILUC can be avoided if productivity, crop choices, land use choices and so forth can be successfully optimized.

Table B. Comparing the scenarios mapped out by Searchinger and Dale

Factor	Assumption by Searchinger	ASSUMPTION BY DALE
Elasticity of food demand to price	Food demand is elastic to price but not as much as supply is.	There is no need to eat less, as the demand can be accommodated in better ways
Elasticity of yield to price	Positive and negative yield effects cancel out.	Biofuel demand drives innovation in energy crop agronomy.
Crop choices	Crop choice responds to price, and farmers are somewhat resistant to change.	Farmers choose to grow high-productivity energy crops.
Utilization of co-products	Distillers grains are returned to the feed market, reducing net corn demand by about one-third.	Co-products from energy crop production are used to feed livestock.
Elasticity of area to price	Land use globally responds to price pressures.	Area in the United States increases to meet demand, so no expansion is necessary elsewhere.
Carbon stock of new land	This reflects historical patterns of land use change, including expansion into high-carbon areas.	Only low-carbon-stock land is brought into production—there is no expansion into high-carbon areas.
Conclusion:	ILUC wipes out the carbon savings of corn ethanol (108 gCO_2e/MJ).	The system can provide both food and fuel without ILUC emissions.

In these two competing visions of biofuels production, different expectations for the six determinants that have been identified makes the difference between (a) projecting that U.S. biofuels policy will cause substantial net GHG emissions and (b) believing that a massive expansion of bioenergy use is possible with no ILUC at all. Other models and scenarios tend to give answers somewhere between these extremes

ES.IV. ILUC—AN ILLUSTRATIVE CALCULATION

Having identified six factors that determine the size of indirect land use change emissions, it is possible to construct a simple model of how each contributes to overall land demand, and consequently ILUC emissions. The ICCT model that is utilized here for the case corn ethanol is explained in more detail in section 4.2 and documented in Appendix C. While this model is useful for exploring the dynamics of ILUC, it is not intended as an alternative to economic models. It cannot capture the complex interlinkages between sectors and regions that are represented in equilibrium economic models. Rather, it can

6 The 2008 paper by Searchinger et al. has been extensively discussed, both critiqued and defended, since its publication. For example, responses including a critical letter by Michael Wang and Zia Haq and the subsequent riposte by Timothy Searchinger are available from the Science website (http://www.sciencemag.org/content/319/5867/1238.abstract).

be thought of as an illustrative tool to sensitivity analysis based on simple linear approximation The illustrative ILUC value for corn ethanol of 30 gCO_2e/MJ (with a 30-year 'amortization', i.e., spreading the emissions effects across a 30-year period) is achieved by calibrating the illustrative model to the Air Resources Board's regulatory modeling for the California Low Carbon Fuel Standard (LCFS); it is not a result from applying the model (although it indicates the sort of assumptions that would be consistent with an ILUC value of 30 gCO_2e/MJ).

As a starting point for the illustration, consider what the ILUC associated with corn ethanol production would be if all of the biofuel feedstock came from converting new land at average yields and typical carbon stocks. With a global average yield of four metric tons of corn per hectare, and assuming that on average 95 tons of carbon are lost per hectare following land conversion, this would result in an ILUC emissions intensity of 190 gCO_2e/MJ, so high that biofuels policies could never deliver carbon savings compared to fossil fuel. However, as the six factors are taken into consideration, the expected ILUC emissions are reduced.[7]

1. **Food consumption falls, reducing ILUC by 30 percent**
   When prices increase, there is normally some response on the supply side (increasing agricultural production) but also a response on the demand side (people eat less, drink less, buy less new cotton clothing, etc.). The elasticity of demand is assumed to be a bit lower than the elasticity of supply; still, the fall in consumption reduces ILUC emissions intensity by 30 percent, to 135 gCO_2e/MJ.
2. **Yields change, reducing ILUC by 15 percent**
   On the supply side, feedstock is boosted by improved yields as farmers respond to higher prices by increasing productivity. The ICCT model has yield contributing one-third of the supply response and area increase giving two-thirds. At the same time, yields on new areas of land brought into production will typically be lower than average. The overall effect of yield changes is to reduce ILUC emissions intensity by 15 percent, to 115 gCO_2e/MJ.
3. **Crop choices change, reducing ILUC by 40 percent**
   U.S. corn yields are higher than the global average. It is assumed that much of the agricultural expansion will happen in the United States, increasing the average yield for land where expansion occurs. The model also allows for crop choices to change throughout the agricultural system as prices adjust, assuming that this dampens total demand for new land by 20 percent. Overall, ILUC emissions intensity is reduced to 70 gCO_2e/MJ.
4. **Co-products reduce ILUC by 40 percent**
   The initial calculation ignores co-products. Nearly 40 percent of the edible content of corn is returned to animal feed markets as distillers grains. In the United States, distillers grains primarily displace feed corn on a one-to-one basis, so this effectively reduces feedstock demand and hence ILUC emissions intensity by 40 percent, to 40 gCO_2e/MJ.
5. **Elasticity of area to price**
   In this illustration, the elasticity of area to price is already 'turned on' when the emissions were calculated assuming that all new production came from new land, so it cannot be switched on sequentially like the other factors. The

7 All 'interim ILUC factors' are rounded to the nearest 5 gCO_2e/MJ, and percentage ILUC reductions rounded to the nearest 5% to deliberately reduce the precision of this high-level illustrative analysis.

important question is how area elasticity compares to the food consumption and yield elasticities. Area elasticity is therefore missing from Figure D, but Figure E illustrates how reducing or increasing the area elasticity would affect the ILUC emissions.

6. **Land expansion tends to avoid higher-carbon biomes, reducing ILUC by 30 percent**
   The baseline of 190 gCO_2e/MJ assumes that expansion affects land types more or less randomly. If, however, one assumes that farmers will prefer grasslands to forests and recognizes that the highest-carbon habitats are outside the United States, whereas it is expected that a larger percentage of land expansion will occur within the United States, the emissions will be lower than the global average. Here, the model assumes that the carbon stock of the average parcel of converted land is only 70 percent of the world average. The resulting ILUC emissions intensity, after all six of the factors are incorporated, is 30 gCO_2e/MJ.

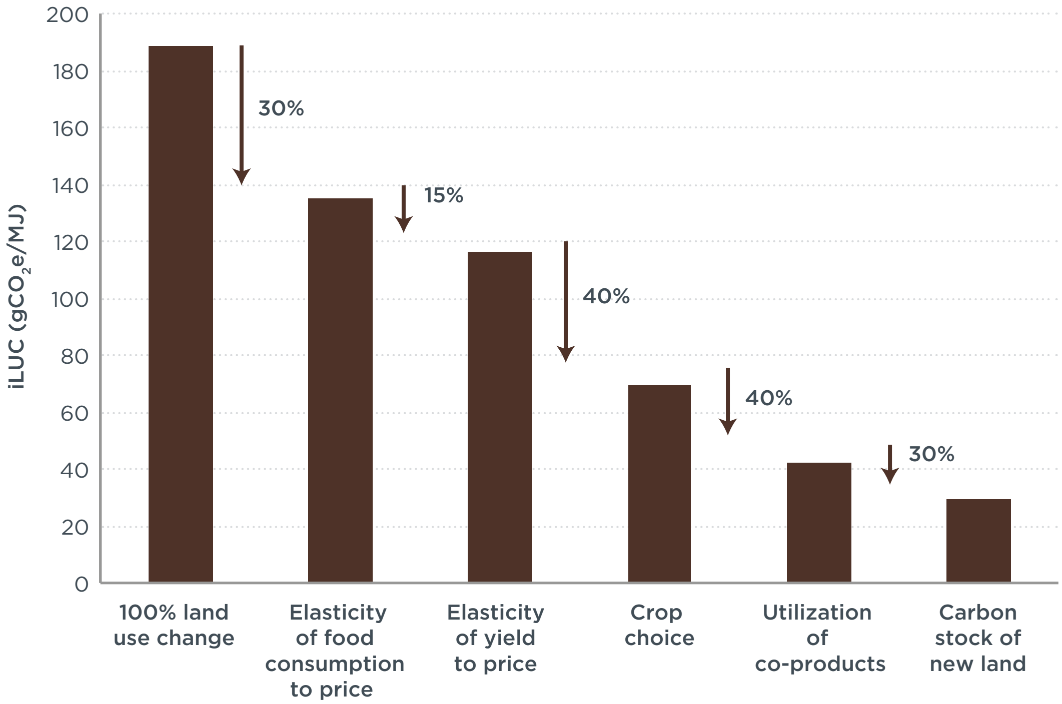

Figure D. Illustration of the importance of various factors in determining the ILUC emissions intensity for U.S. corn ethanol

The reductions in expected ILUC emissions intensity as each factor is considered are shown in Figure D. In this case, all of them are important in determining the final outcome, with the return to the market of co-products as feed and the assumptions about crop choice making the largest difference.

As well as providing an illustration of how each parameter affects ILUC, the simple model can also be used to provide an indication of the importance of each. Working around the 'central case' estimate for corn ethanol ILUC given above, a best case and a worst case can be outlined for the ILUC implications of each factor. These are based on the range of parameter values reported in the literature, combined with the ICCT's expert judgment. Figure E shows four ILUC possibilities for each characteristic. When the central cases are used (brown dots), the overall ILUC is 30 gCO_2e/MJ. The blue bars show what the ILUC estimate would be if a given determinant were ignored entirely and the others were held at their central assumption. For example, not considering the utilization of co-products would shift the estimated ILUC upward from 30 gCO_2e/MJ to about 50 gCO_2e/MJ. Finally, the top and bottom of the range marked by the error bars represent the ILUC emissions for best and worst cases. Low and high assumptions for land carbon stock, for instance, could vary the estimate ILUC from 10 to 100 gCO_2e/MJ. The variation from best case to worst case reflects the importance of a particular determinant to ILUC emissions but also speaks to how differently each is treated in existing models and studies.

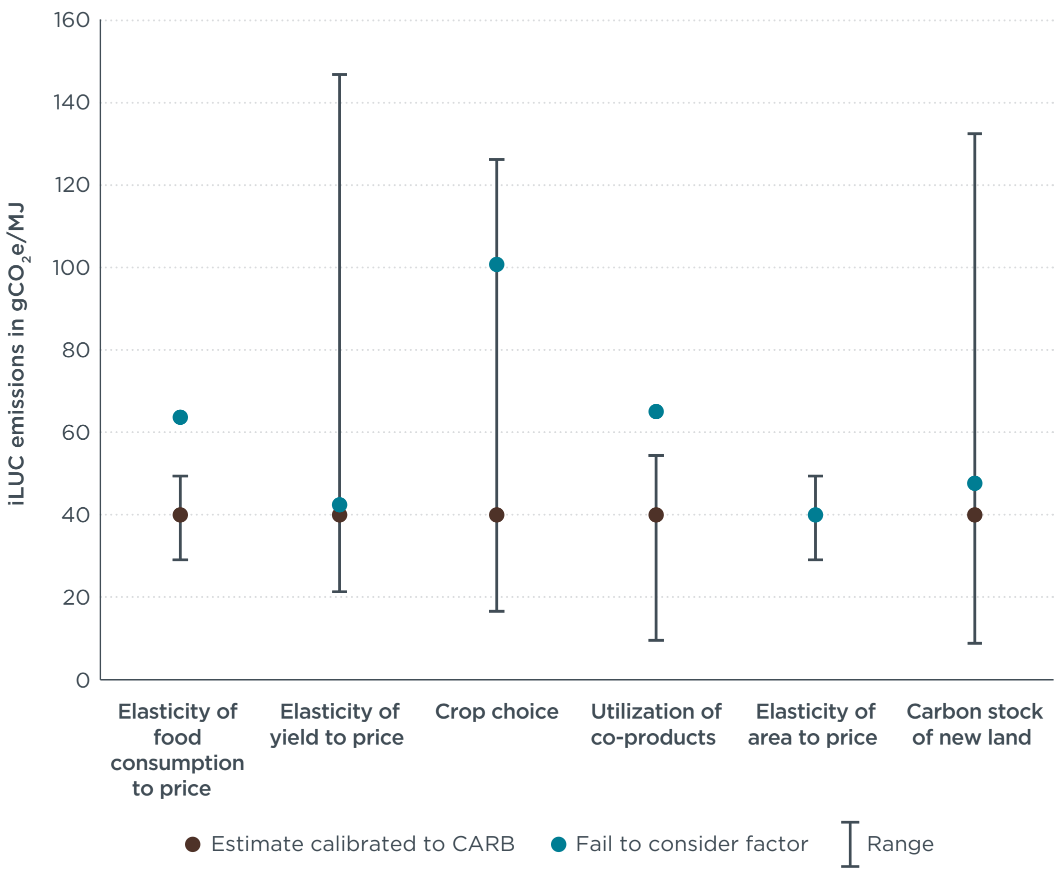

Figure E. Illustrative model of how more or less favorable assumptions for each parameter could affect ILUC results for U.S. corn ethanol

Note: The blue bars for 'Fail to consider' represent the level of ILUC emissions intensity expected if one parameter is completely eliminated, and thus can sometimes fall outside the range of plausible values.

The largest ranges are those for yield elasticity, crop switching, and carbon stored in soil and vegetation. In the case of yields, this wide range is attributable to the sharply differing positions presented in the literature. ILUC is high if one assumes that there is no direct response of yield to price but that yields on land at the margin of production (the new land used to supply extra feedstock) are much lower than average yields; it is low if the yield elasticity is as strong as the area elasticity and if new land under cultivation achieves average productivity. For carbon storage, the range illustrates how much difference it makes whether expansion affects high- or low-carbon ecosystems. Crop switching's broad range reflects that this phenomenon has not been widely discussed in the literature, but different models ascribe widely varying contributions to these effects.

Where the ranges are narrower, this does not necessarily mean that these concerns are less important. Food consumption reduction, for instance, is central to reducing ILUC, but there is relative consensus in the literature that food consumption accounts for between 20 and 50 percent of feedstock (though there are modeling exercises that go outside this range).

These ranges serve as a valuable reminder that the outputs from a model are only as good as the inputs. With this simplified model, combining all the worst-case assumptions for corn ethanol would give an ILUC emissions intensity projection of nearly 3,000 gCO_2e/MJ, while combining the best-case assumptions would actually give a credit (improvement) of nearly 30 gCO_2e/MJ. The fact that different assumptions give different answers should not be surprising, nor should dissent in the literature be understood to imply that model results have no value. The lesson from this exercise is that any modeling result maps out a scenario for how the world responds to a biofuel mandate and that only by understanding the assumptions that go into building that scenario can the reasonableness of a model's estimates be judged.

ES.V. UNDERSTANDING THE FACTORS

ES.V.i. Demand change: Elasticity of food demand to price

When demand for food-based biofuels goes up, so do the prices of food commodities. One of the ways people respond to higher prices is by consuming (i.e., eating) less. Econometric work by Roberts and Schlenker (2010) suggests that at least a third of the additional feedstock for biofuels will be made available in this way on the demand side, with another third to a half coming from yield improvement and expansion of the area under cultivation. Most economic modeling exercises show clear food commodity price increases and reductions in consumption as biofuel mandates expand (Figure F). If feedstock comes from reduced consumption instead of land expansion, this will of course reduce ILUC emissions.

'Food vs. fuel' is a contentious topic, and a variety of international organizations, experts, and antipoverty campaigners have called for biofuel support to be curtailed or ended to reduce pressure on food markets.[8] The U.S. Department of Agriculture's Economic Research Service (2003) finds that poorer people's food consumption declines faster when prices rise than that of rich people. Many voices in the biofuels industry have responded by arguing that competition between food and fuel is fictitious or exaggerated. The reality is that, even though biofuels are not by any means the only thing affecting food markets, the weight of evidence (including analysis of the 2008 food price spikes) finds that biofuels increase both prices and price volatility. Both of these effects will tend to have negative impacts on welfare.

8 For example, the UN Food and Agriculture Organization, the International Fund for Agricultural Development, the International Monetary Fund, the Organisation for Economic Co-operation and Development, the UN Conference on Trade and Development, the World Food Program, the World Bank, the World Trade Organization, the International Food Policy Research Institute, and the UN High-Level Task Force on the Global Food Security Crisis (2011); the Institute for Agriculture and Trade Policy and the Global Development and Environment Institute (2012); the UN Food and Agriculture Organization High Level Panel of Experts on Food Security and Nutrition (2013); the UN Special Rapporteur on the Right to Food (2011); Laborde (2011b); the Financial Times (2008); World Bank President Robert Zoellick (2008).

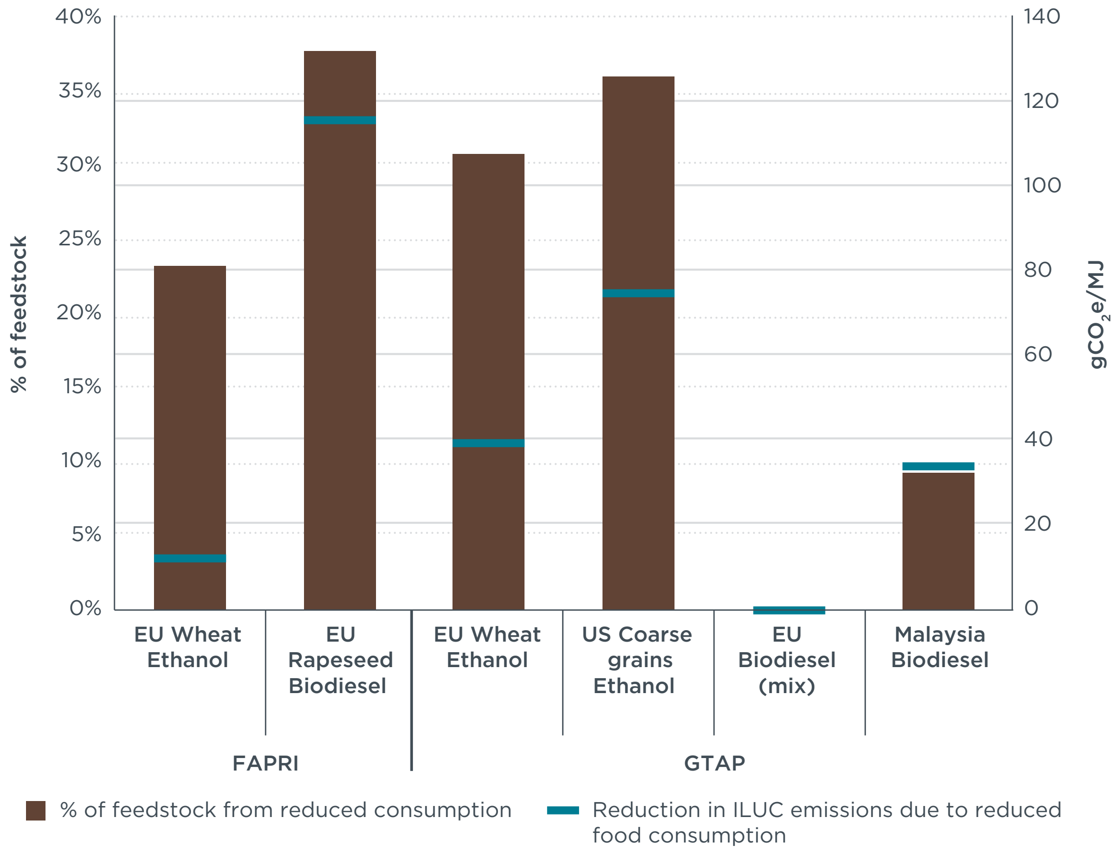

Figure F. Relative importance of reduced food consumption in different GTAP and FAPRI scenarios*

* Percentage of feedstock from reduced consumption (left axis) and an approximation of the additional carbon emissions that would have resulted had this feedstock come from land expansion instead (right axis; based on Edwards, Mulligan and Marelli (2010), using a 20-year spread, or "amortization")

Increased food prices can deliver economic benefits to farmers, but because most poor people (even including many small-scale farmers) are net purchasers of food, on average increased food prices affect the poor negatively. Modeling that compares the economic costs and benefits of increased food prices suggests that, globally, tens of millions of people may be pushed below the poverty line because of the current generation of biofuel policies. While poverty increases are not an intended outcome of these policies, they might be deemed an acceptable consequence of an effective carbon reduction policy, or they could be compensated for through other policy mechanisms.

ES.V.ii. Productivity change: Elasticity of yield to price

Historically, the standard agricultural yield trend has been one of fairly steady linear growth. For individual crops and regions the picture might be different, but in general there seems little reason to think that the rate of yield growth in the next decade will be radically different from what has been observed in the past. Climate change effects on yield seem unlikely to be large in the period to 2020; thus, ignoring them in current ILUC modeling is probably reasonable.

Could the introduction of biofuel mandates affect yields? Microeconomics suggests that if revenues improve, farmers will have incentives to spend more to boost yields, for instance, with extra fertilizer or by adopting new technologies. On the other hand, when farmers cultivate new land, it is likely that this will be less fertile than currently farmed plots (or else it would be in cultivation already), in which case average yields might fall. While historical data clearly show that consumption and land use respond to demand, there is currently no statistically convincing historical evidence for a yield response or a price-led increase in the rate of innovation. There is also no consensus on the relative productivity of land at the margin of production, although values of between 70 and 100 percent of the average seem likely.

While there is a lack of compelling evidence for a strong yield response to price hikes, it is nevertheless included in almost all ILUC modeling—several modelers have also included parameters for lower yield on new areas farmed. These yield effects have been important determinants of modeled ILUC emissions intensity, with the price-induced yield effect generally being the larger and helping to reduce ILUC estimates. The EU's Joint Research Centre (Edwards, Mulligan, and Marelli 2010) provides a comparison of the importance of price-induced yield effects in reducing land demand in different modeled scenarios (Figure G). Yield gains cut emissions intensity by more than 30 gCO_2e/MJ in many cases.

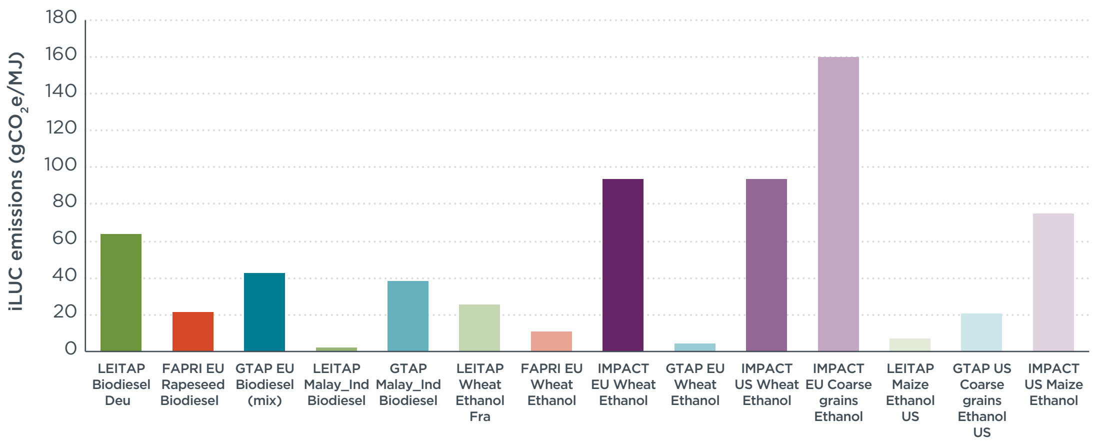

Figure G. Decomposition of importance of price induced yield change in models (Edwards, Mulligan and Marelli, 2010)

ES.V.iii. Productivity change: Crop choices

Yields vary between (and within) regions, and so the location of agricultural expansion is important. For instance, corn yields are much lower in Africa than in America. There are also differences between yields of different crops: for instance, oil palm has a much higher vegetable oil yield than sunflower.[9]

The location of expansion depends to a great extent on how well connected the agricultural markets in different countries are. If the price signal from increased U.S. demand for corn were much stronger domestically than globally, production increases would largely be confined to America itself. If in contrast there is a single world market for corn,

9 Of course, land area requirements do not always scale with land use change emissions—arguably, one hectare of palm expansion will result in larger emissions than several hectares of sunflower expansion; see section 3.6.4.

additional production would likely be more widely distributed for import to the United States (or to compensate for reduced U.S. exports).

Figure H[10] shows how assuming land use change in areas with yields lower or higher than the world average can change the net demand for additional land. In the cases of U.S. corn and EU wheat, there tends to be a saving (because expansion is assumed to occur largely in those regions, which have high yields), but in the case of rapeseed biodiesel, expansion is predicted in areas of lower than average oilseed yield, increasing ILUC.

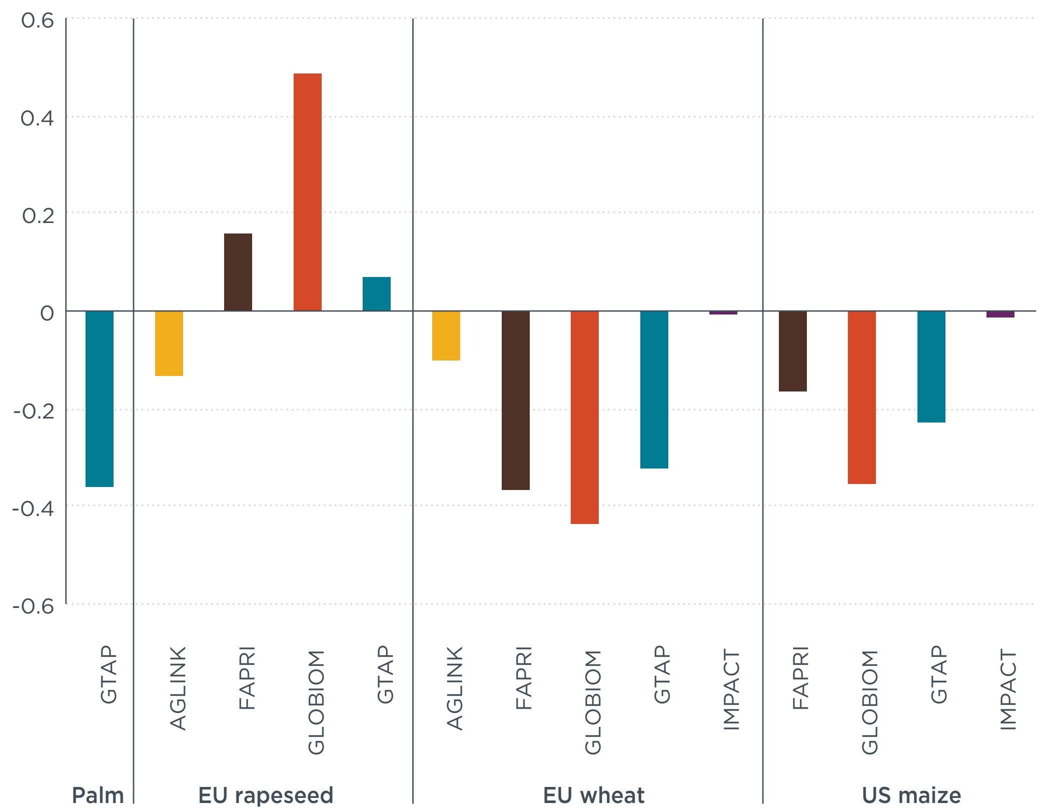

Figure H. Effect on modeled land use of expanding crops in areas with yields higher or lower than the world average

Source: Based on methodology outlined in Witzke (2010)
Note: Area saved (shown as negative values) or required (positive values)

There could also be switches in crop choices on existing agricultural land. In principle, if it were possible systematically to replace lower-yielding crops with higher-yielding ones, it could substantially reduce the requirement for land expansion. For instance, rapeseed has a higher average yield in Europe than sunflower seed, so shifting cultivation from sunflowers to oilseed rape would seem to be one option to boost the average yield of agriculture in Europe as a whole. Having said this, average yield statistics could mask good reasons for farmers to grow sunflowers in the first place, and therefore it could be naïve to assume that a wholesale shift would be desirable or even possible. These types of switching effects are allowed in models but have not been widely discussed in the literature, and their role in reducing ILUC may have

10 Based on a modification of the decomposition analysis proposed by Witzke et al. (2010); see Appendix A.

been overestimated in cases where models are not able to incorporate the agronomic reasons for current land allocations.

ES.V.iv. Productivity change: Utilization of co-products

When part of a biofuel crop cannot be turned into liquid fuel, rather than being discarded, it will often be sold (with or without further processing) as a co-product. For instance, when corn is fermented to produce ethanol, about a third is left over as distillers grains. By displacing other animal feed from livestock diets, co-products can provide an 'ILUC credit' to biofuels.

In the United States, it has generally been assumed that distillers grains with solubles (DGS) will largely displace feed corn. Some studies using feed trials have suggested that one metric ton of DGS will displace more than one ton of corn, but it seems likely that these results are an artifact of the test methodology and would not be realized in actual animal diets. There is a substantial literature on the nutritional value of co-products (see section 3.4.2), and it seems likely that in reality co-products will displace a more complex mix of ingredients. Still, an assumption of a 1:1 replacement of feed corn is likely to be a reasonable approximation for the purposes of modeling.

In Europe, much discussion has focused on the higher protein content of DGS and oil meals compared to wheat. It has been argued that in the European livestock sector co-products will substantially displace protein-rich soy meal, currently imported from Latin America. Because soy has a lower yield than either wheat or corn, and soy production in Brazil has been linked to Amazon deforestation, there may be a case for a larger ILUC credit for DGS in Europe than in the U.S. The nutritive value of feed can be well quantified, and thus the importance of protein content can in principle be captured in economic modeling. For instance, modeling for the International Food Policy Research Institute by Laborde (2011a) explicitly accounts for the protein content of feeds. While it is certain that co-products are fed to livestock and displace other feed products, the net effect on ILUC emissions also depends on any overall changes in the feed market. The modeling by Laborde suggests that adding large quantities of protein feed into the market could cause a protein 'rebound effect'—with overall protein consumption rising as prices are pushed down and little overall change in global soy demand.

ES.V.v. Land use expansion: Elasticity of area to price

Sometimes, especially when talking about demand reduction, yield increase, or by-product use 'reducing net land demand,' it can sound as if indirect land use change is some sort of remainder term: if biofuel feedstock cannot come from yield growth or demand dampening, there is grudging acceptance that land needs to expand. In fact cultivated area expansion is an active response that happens in parallel to these other responses, so that even if food demand and yield are both affected by price, total cultivated area could be still *more* price sensitive. In that case, one would expect much or most of the extra feedstock to come from land expansion.

The basic economic argument is that, as prices rise, it should become profitable for farmers to cultivate additional area. Several historical econometric studies find statistically significant area response to prices, though this varies from region to region. Agriculture in regions dominated by strong government intervention, or where there is simply less opportunity to expand, is expected to be less price responsive. Berry and Schlenker (2011) find that agricultural area in Brazil (with its economy oriented to agricultural exports and its large potential for expansion) is strongly price elastic, whereas in India, with strong government intervention and land at a premium, there is generally not a statistically significant relationship.

The Joint Research Centre study (Edwards, Mulligan, and Marelli 2010) allows for comparison of the extent to which demand for new biofuel feedstock drives demand for new land in various modeled scenarios. Figure I shows that in comparing 'gross land demand' (the amount of land needed at world average crop yield to produce all of the total required biofuel feedstock) with 'net land demand' (the amount of land expansion that happens in the model), the results vary widely. In some cases, the models have nearly one hectare of land expansion for every hectare of demand. In others, there is only one-tenth as much.

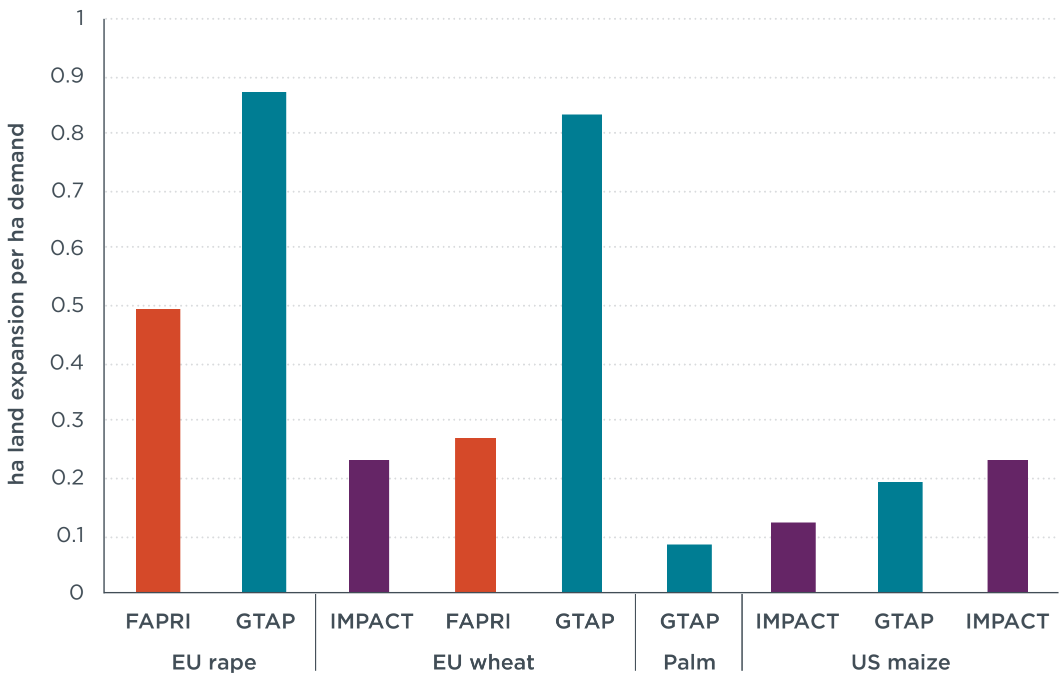

Figure I. Estimates of the hectares of land use expansion required per hectare of gross biofuel land demand

Source: Edwards, Mulligan and Marelli (2010)

ES.V.vi. Emissions implications: Carbon stock of new land

Knowing the total number of hectares of land use change is not enough to work out ILUC emissions—it must be linked to the carbon consequences of converting those hectares, and the carbon stock of land (in both biomass and soil) can vary widely.

When land is brought into cultivation, almost all of the carbon in plant biomass as well as typically about 30 percent of soil carbon, will be released to the atmosphere as CO_2.[11] An important special case is the drainage of peat soils for the cultivation of palm oil. When peat is drained, it begins to decompose immediately, and over the first 20 years after drainage, emissions from peat soils in Southeast Asia average 105 metric tons of CO_2 per hectare per year (Page et al. 2011a). Based on current trends, at least a third of future oil palm expansion is expected to occur on peatland, with enormous

11 After forest clearance, on average about 10 percent of wood is likely to be used for timber or for other purposes in developed countries and at most 3 percent in the developing world (Searle and Malins, 2011).

emissions implications. In addition to the release of carbon into the atmosphere upon conversion, there may be a loss of carbon sequestration that would have occurred naturally had conversion not taken place. For example, in average abandoned cropland in forest biomes[12] in Europe, 240 tonnes of CO_2 per hectare of carbon sequestration through forest regrowth could be missed over 30 years.

As well as understanding the carbon consequences of any given land use change, calculating ILUC requires predicting which types of land are likely to be converted. One approach is to look at historical trends, using either reported data or techniques such as satellite identification.[13] An alternative/complementary approach is to use economic modeling based on land rents. In that case, the relative revenue available from cropland versus managed forest versus pastureland determines which land use changes occur. A third alternative is to use a more sophisticated assessment of land characteristics to determine where land conversion is likely. The Joint Research Centre (Edwards, Mulligan, and Marelli 2010) proposes a spatial allocation system for assessing which land use changes might be most likely and assigning emissions consequences to them. Systems relying on historical analysis are sound provided that it is reasonable to believe that future land use change will echo past patterns. Systems based on economics or agricultural suitability assessments can give reasonable results provided that they adequately capture real-world drivers of land use decision-making. Certainly, whichever way models attempt to predict the areas where land use change will occur, there is no reason to believe that agricultural expansion will uniformly occur on land with low carbon stocks—both history and economics support the expectation that a mix of ecosystems, from peat bogs to forests to grasslands to pasture to idle cropland, will be affected.

ILUC models generally use a set of land use change emissions factors combined with a model for which land types will be converted. For comparison, the Joint Research Centre study (Edwards, Mulligan, and Marelli 2010) recalculated ILUC emissions from several models assuming a single flat rate of land use change emissions (147 metric tons of CO_2 equivalent per hectare), with an additional allocation for peat emissions where appropriate—the difference is shown in Figure J.

12 Land that would be expected eventually to revert to forest in a natural state.

13 Notably, a study using MODIS imagery by Winrock International (Harris, Grimland, and Brown 2009), which has been used in both the U.S. EPA's FAPRI-FASOM ILUC analysis and the International Food Policy Research Institute's MIRAGE modeling for the European Commission; also, for instance, Miettinen et al. (2012) used satellite imagery to examine peat loss.

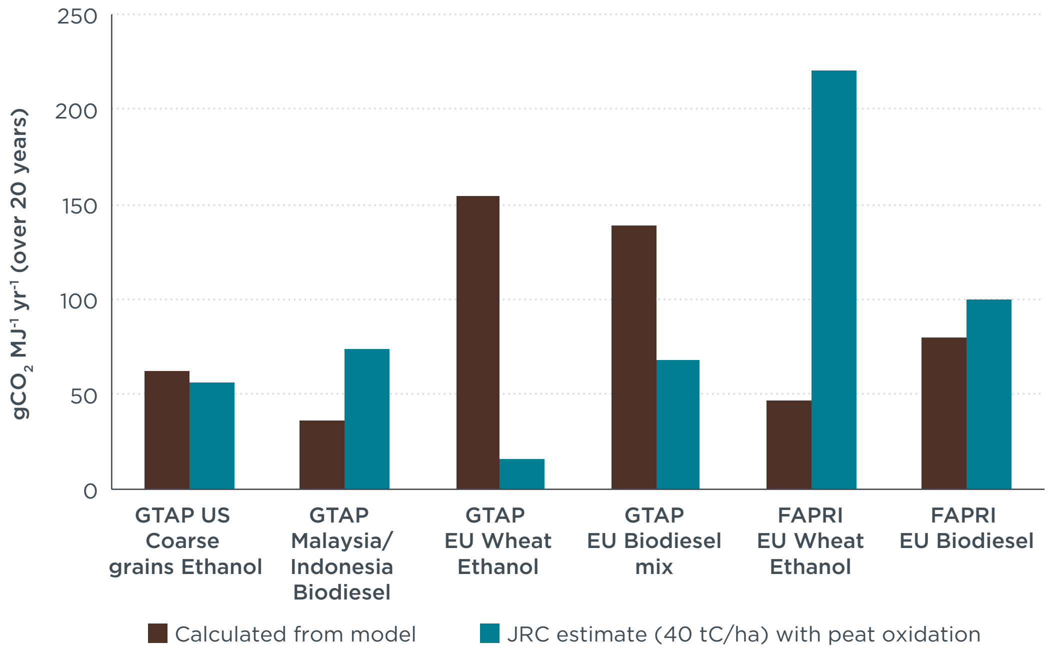

Figure J. Comparing ILUC emissions intensity derived by assuming a single average emissions rate versus differentiating land types

Source: Edwards, Mulligan, and Marelli (2010, p100, Figure 23)

ES.VI. CONCLUSIONS

Since 2008, indirect land use change (ILUC) has become a key concern for policymakers. There are six key factors (elasticity of food consumption to price, elasticity of yield to price, crop choice, utilization of co-products, elasticity of area to price, and carbon stock of new land), that combine to determine how large ILUC emissions will be, and researchers have used a variety of different models and modeling approaches in the attempt to estimate ILUC emissions from various biofuel pathways. ILUC modeling is not an exact science—indeed, because it is built on the construction of scenarios and counterfactuals, it can never be an exact science—and the modeling to date has given a wide range of results. The ILUC modeling outputs reflect the interplay of myriad real-world decisions, simplified and aggregated to allow them to be expressed in mathematical form, and changing any of the inputs, functional forms, or assumptions of the system would affect the outcomes.

The absence of a single globally agreed framework for ILUC analysis is not, however, a reason to dismiss the results or their importance for policy. There is a long tradition of using economic modeling exercises to inform decisions in many areas of government (e.g., structural adjustment policies, international trade, public finance, agriculture, income distribution, and energy and environmental policy [Devarajan and Robinson 2002]). Biofuels policy must acknowledge the ILUC results if regulatory instruments are to be effective in mitigating atmospheric carbon buildup. One policy approach to incorporating land use change effects involves working with the models to establish 'ILUC factors' in regulatory emissions accounting. This has been done in the RFS2 and the LCFS, providing a value signal to suppliers about which biofuels are expected to deliver the largest emissions savings. The modeling results can also be used to guide more qualitative decisions, such as the European Commission's 2012 proposal to report

ILUC and to incentivize biofuels that do not require land use. At the moment, however, fuel policies in China, India, Canada, Mexico, Japan, Australia, and Brazil are promoting biofuels with no accounting or reporting of indirect land use change emissions.

The evidence is clear that mandates for crop-based biofuels drive land use change, causing significant ILUC emissions. Recognizing the importance of ILUC puts a question mark over the carbon savings that have previously been assumed for many biofuels. Nevertheless, there is a positive side to the results of ILUC modeling because 'reverse-engineering' the logic of ILUC points to opportunities to reduce emissions from biofuels production and from agriculture in general. For instance, the paper "Biofuels Done Right" by Bruce E. Dale et al. (2010) shows that if the response to biofuel demand could be limited to the United States only, with advantageous crop switching, expansion of cultivation targeted to idle land with relatively low carbon stocks, and innovations in livestock nutrition to allow cellulosic biofuel co-products to be used as feed, then a large-scale U.S. biofuels industry could be possible without major ILUC emissions (see also Table B). Biofuel mandates alone would not be enough to make this happen, but there is an opportunity for policymakers to work with the farming industry to introduce a new generation of more sophisticated regulatory guidance to help steer agriculture toward a more beneficial model of development.

More broadly, this assessment suggests that a change is warranted in international conventions for biofuel carbon accounting. For example, the Kyoto Protocol and the European Emissions Trading System account biofuels as 'carbon neutral' (meaning that they have zero carbon emissions), but the evidence shows that for a biofuel truly to have low carbon emissions requires that ILUC is minimized by improving the efficiency of the agricultural system, bringing fertile land with low carbon stocks into production, or reducing the consumption of food and feed.

For each of the six factors identified here, further research would help narrow the range of ILUC results and progressively increase confidence about the magnitude of ILUC for each feedstock. On food, there are still many questions about the way biofuel demand affects price volatility and whether biofuels policy could be structured to dampen rather than reinforce price spikes. Direct monitoring of yield at the margin of production would help refine the ILUC models. Crop switching is an area in which several models suggest large effects, but it has not been extensively studied in this context. As the supply of co-products becomes routine in the feed sector, there is a growing volume of data available to confirm or revise expectations for feed displacement. On area elasticity and carbon stocks, stakeholders have argued repeatedly that there are large areas of low-carbon land that will preferentially be converted to supply biofuels—this needs to be tested.

There are good biofuel pathways—but it is incumbent on both government and industry to provide convincing evidence that biofuel mandates can be reasonably expected to deliver the carbon savings that are claimed for them. Regulations that use credible modeling to establish ILUC factors for emissions accounting can target incentives to the biofuels most likely to deliver carbon savings. At the same time, ILUC can be avoided by promoting pathways such as biofuel from low-value residues and from perennial grasses expanding onto low-productivity land.

1. INTRODUCTION

According to the International Energy Agency (IEA), between 2000 and 2010, global biofuel supply went up by a factor of more than five (Figure 1.1), driven largely by strong growth in the use of corn ethanol in the United States and, on a lesser scale, by European biodiesel demand and Brazilian sugarcane ethanol expansion. For policymakers, there was a time when biofuels represented an apparent triple win—reducing climate-forcing carbon dioxide emissions, improving energy security by diversifying the energy supply, and supporting farmers by increasing crop demand and prices.

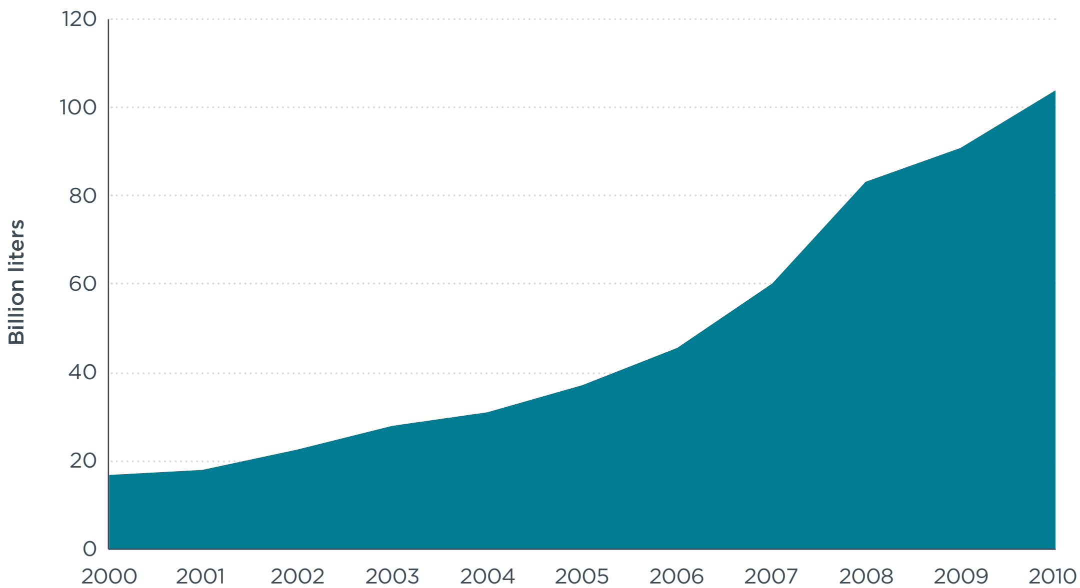

Figure 1.1 Global biofuel supply, 2000–2010

Source: International Energy Agency, 2010

More recently, however, the brief consensus that biofuel support was a good policy strategy has been broken as the focus has shifted toward the potential drawbacks and the benefits have been questioned. One commonly expressed concern has been environmental sustainability—does the monoculture agricultural model typically used to supply biofuel feedstock have negative consequences for biodiversity, air and water quality, and soil erosion? Will biofuels plantations be allowed to expand onto forests, wetlands, or other high-carbon-stock ecosystems? Both European and American biofuels legislation have been expanded to provide some response to these concerns,[14] but regulatory requirements still fall far short of the broader sustainability assessments required by initiatives like the Roundtable on Sustainable Biomaterials or considered by the Global Bioenergy Partnership. There have also been concerns on the social side, especially with regard to developing countries, where some commentators have argued that food security has been adversely affected, that agricultural expansion is undermining land rights and harming existing communities, and that plantation workers in many cases suffer from low wages and poor working conditions.

14 The European Renewable Energy and Fuel Quality directives and the U.S. Renewable Fuel Standard contain some basic sustainability protections for some high-carbon ecosystems and highly biodiverse ecosystems.

These issues are important, but there has also been a more fundamental question: What are the implications of taking the biofuel feedstock out of the global market? The current boom in biofuels production is not, in general, supplied by farms specifically committed to growing biofuel feedstocks. Rather, biofuels producers are buying feedstock from a market in which they compete with other uses, notably food and fodder. It is not immediately obvious whether the area cultivated will expand to supply this extra feedstock, whether one can expect a reduction in food consumption, or whether the system will be able to become more efficient (higher yield) to expand supply. In reality, some combination of these effects is expected to occur. Each will have consequences, however, and in a world where it is generally agreed that the pressure on natural resources will only increase in the coming decades, the benefits of biofuel support could be significantly reduced depending on how the additional demand is met. The discussion has been largely channeled into two interconnected debates: 'food versus fuel,' about whether price increases caused by biofuel demand will increase food insecurity, and the indirect land use change conundrum, whether land expansion to supply biofuels will cause so much carbon to be released as a result of land conversion that it cancels out the benefits of replacing fossil fuels.

1.1 INDIRECT LAND USE CHANGE

Mandates create additional demand for biofuels. This demand must be met either by reducing use of biofuel feedstocks in other sectors, or by increasing their overall production. One obvious way to increase agricultural production is to bring new land under cultivation—if and when this happens in response to biofuel demand, it is called *indirect land use change* (ILUC).

Indirect land use change versus direct land use change

The distinction between direct and indirect land use change (DLUC and ILUC) can be a source of some confusion. The distinction is as follows:

» Direct land use change happens when a particular parcel of land is converted to use for biofuel production, for instance, if an area of forest in Malaysia is chopped down and replaced with oil palms, and the oil is then supplied to a biodiesel plant. However, categorizing a land use change as direct says nothing about whether that land use change would have happened in a given period if there had not been an increase in biofuel demand. Perhaps that Malaysian forest tract might have been felled to make space for food production anyway.

» Indirect land use change is the set of land use modifications that would not have happened without a marginal increase in biofuel demand. For example, biofuel demand raises palm oil prices by 20 percent in Malaysia, and this leads palm oil producers generally to clear more forest to take advantage of improved profit margins.

» Modeling indirect land use change is about comparing a world with a "baseline" level of biofuel demand to a world with some defined increased level of demand, the "biofuel scenario". The baseline and biofuel scenarios can either be modeled 'now', in which case the baseline should reflect the world as it is, or at some time in the future, in which case the baseline and biofuel scenario represent two different future projections.

- Not all direct land use change is part of indirect land use change and vice versa. Many examples of direct land use change may have been likely to happen anyway in a given time frame, owing to expanding agricultural demand outside the biofuel sector. In that case they would not be part of the indirect land use change. It would be possible in principle to have no biofuel feedstock actually grown on newly converted land, in which case one could have only indirect land use change and no direct land use change.
- Economic models do not attempt to link specific patches of land to specific end uses for commodities and in fact are incapable of doing so. Thus, indirect land use change analysis says nothing about which specific parcels of land are supplying the feedstock actually used for biofuel production, and does not attempt to identify direct land use changes.

The land use consequences of biofuels play a critical role in the evaluation of their effects on global greenhouse gas emissions. The more grassland or forest that would need to be plowed up to replace feedstocks previously used for food and fodder, the more carbon stored in plants and soils would be released. In some cases, there may also be missed opportunities for continuing carbon sequestration (notably, when immature forest is cleared or when idled land that would otherwise be reforested is brought back into cultivation).

A number of studies have tried to quantify or otherwise assess the scale of the carbon consequences of this indirect land use change. The first peer-reviewed effort to measure the possible magnitude of indirect land use change emissions was made by Searchinger et al. (2008). Timothy Searchinger's team used the partial equilibrium economic model[15] of world agriculture developed by the Food and Agricultural Policy Research Institute (FAPRI) to estimate how much land use change might be expected from meeting U.S. corn ethanol mandates. It then assigned likely carbon emissions to this amount of land use change by using average land carbon content estimates from the Woods Hole Oceanographic Institution. The modeling suggested that the carbon emissions intensity of land use change would be 104 grams of carbon dioxide equivalent per megajoule (gCO_2e/MJ), larger than any potential carbon savings from reduced fossil fuel use, and therefore the team argued that corn ethanol programs would increase rather than decrease carbon emissions.

The Searchinger study led to a surge of interest in the question of whether indirect land use change would overwhelm any carbon benefit from biofuels and resulted in government, academia, business, and the nonprofit sector producing a variety of follow-up reports. This report does not aim to be a comprehensive literature review, so it will not cover all of them, but here is a brief overview of some of the most important results:

1. 2007/2008: Uwe Fritsche's team at the Öko-Institut proposes the idea of 'ILUC factors.' In the original concept, a simplified system for estimating possible ILUC (based largely on crop yield) would be used to assign ILUC emissions values to specific feedstocks. More recently, the term 'ILUC factor' has become

15 ILUC modeling approaches are explained in further detail in section 1.2

the customary way to denote any system in which ILUC emissions calculated with a model are added to the life cycle analysis of biofuels production.[16]

2. 2008: The UK government's Renewable Fuels Agency publishes the *Gallagher Review of the Indirect Effects of Biofuels Production.* This review paper (and supporting studies) examined various issues surrounding indirect land use change. The report argued that the economic modeling tools available at the time were inadequate to predict ILUC emissions with any certainty but concluded nevertheless that ILUC was a significant problem and that, unless it were addressed, there could be no confidence that biofuels were offering substantial net global carbon emissions reductions, or for that matter any at all.
3. 2009: The Air Resources Board (ARB) of California includes ILUC modeling in the development of its Low Carbon Fuel Standard (LCFS) regulations. The ILUC values are calculated using the Global Trade Analysis Project (GTAP) general equilibrium economic model to project the magnitude of possible emissions from ILUC. Unlike the FAPRI model, which considered only the agricultural sector, GTAP takes in the entire economy. The GTAP modeling, again combined with land carbon content values from Woods Hole, suggests lower ILUC emissions (30 $gCO_{2e/}$MJ) than predicted by Searchinger et al. (104 $gCO_{2e/}$MJ).
4. 2009/10: The U.S. Environmental Protection Agency (EPA) consults on ILUC values calculated using a combination of the global FAPRI model and the Forest and Agricultural Sector Optimization Model (FASOM) for the United States. After extensive comments, the revised FAPRI-FASOM results are included in the Renewable Fuel Standard 2 (RFS2) volumetric biofuel requirement's compliance accounting.
5. 2010: The European Commission publishes several ILUC studies as background documents for a consultation on dealing with indirect land use change. These include a partial equilibrium modeling effort (with AGLINK), a general equilibrium modeling effort (with the International Food Policy Research Institute's Modeling International Relationships in Applied General Equilibrium, or IFPRI-MIRAGE), a comparison of modeling to date (by the European Union's Joint Research Centre [Edwards, Muligan and Marelli 2010]) and a literature review (by the Directorate-General for Energy). Both modeling studies suggest that significant emissions are likely, potentially eliminating any climate benefits from the program. The modeling comparison shows significant emissions consequences in all cases considered, while the literature review emphasizes that uncertainty remains in ILUC assessment.
6. 2010: Throughout the year, ARB runs an Expert Workgroup on Indirect Land Use Change. The group divides into subgroups and generates several reports on fundamental issues regarding ILUC and ILUC modeling.
7. 2010: Tyner et al. (2010) release a revised GTAP ILUC modeling study undertaken for the U.S. Department of Energy's Argonne National Laboratory in

16 There are several ways that the indirect emissions implications of bioenergy production, including ILUC, could be incorporated into life cycle analyses. For the California Low Carbon Fuel Standard, for instance, attributional estimates of "direct" biofuel emissions are added to consequentially estimated ILUC factors. Exploring the wider discussion over how ILUC factors can or should be included in regulation is beyond the scope of this paper. For further reading on carbon accounting for bioenergy, see, for instance, Sanchez et al. (2012) and U.S. EPA SAB (2012).

which they find lower outcomes for corn ethanol ILUC than in the research used for ARB's LCFS (Hertel et al. 2010a).

8. 2011: Plevin et al. (2010) publish an uncertainty analysis of the possible magnitude of ILUC emissions based on a simplified assessment model. They argue that emissions intensity is likely to be on the order of 50 gCO_2e/MJ and that the uncertainty profile has a long rightward tail—that is, ILUC emissions could be very high indeed, meaning that the statistical expected value may be higher than the 'typical' values used by ARB and the EPA.
9. 2011: The European Commission releases updated modeling from IFPRI using the MIRAGE model. The conclusion is that, in the EU, biodiesel is unlikely to offer emissions savings compared with fossil fuel diesel but that bioethanol from cereals or sugars probably holds carbon reduction potential.

It is reasonable to say that, since 2008, a degree of consensus has developed in the scientific and regulatory community that ILUC is a real and significant effect in comparison to the potential emissions savings offered by using biofuels. Still, there remains a great deal of discussion about the magnitude of emissions intensity for individual fuels, with estimates spanning a range from below 10 to above 100 gCO_2e/MJ, depending on feedstock. Some authors even predict negative ILUC effects (additional carbon sequestration), while the very highest estimates go above 200 gCO_2e/MJ. Typically, gasoline and diesel life cycle emissions intensity is around 90 gCO_2e/MJ, so for the highest estimates, ILUC alone would be enough to give a biofuel a higher carbon intensity than the fossil fuel it replaces. Economic modeling has taken a place at the center of the ILUC debate, and it represents the most common approach to trying to quantify emissions. Figure 1.2 gives an indication of the wide variation in ILUC factors found by of modeling approaches that have been applied since the publication of Searchinger et al. (2008). It shows that there is a trend for biodiesel estimates to be higher than ethanol estimates and that this is reflected in the range of ILUC factors proposed by regulatory agencies.

The Joint Research Centre's comparison of economic modeling efforts provides useful insight into the differences between the models. At the most basic level, the report shows that the projected area that would be converted to cropland as a result of a given increase in biofuel demand varies by a factor of three or four, more still if outliers are included (Edwards, Mulligan and Marelli 2010, p. 95, Figure 22). This number (the area of expansion of cultivation required for every unit of increase in biofuel demand) must be multiplied by the average carbon released by each hectare in the process of conversion to determine the expected magnitude of ILUC emissions.

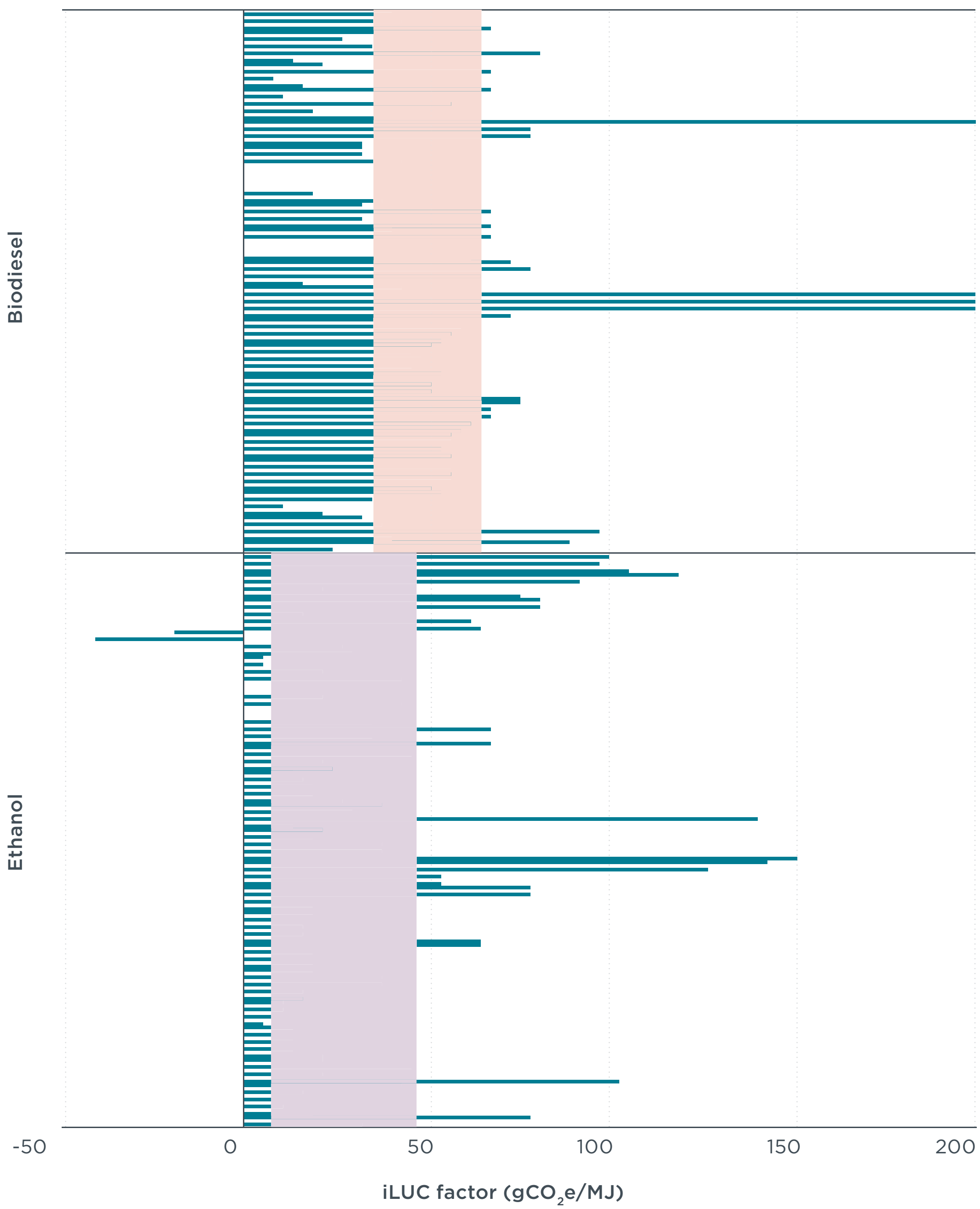

Figure 1.2 Variation in ILUC emissions intensity from different modeling exercises

Notes: The ranges shown for regulatory estimates of ILUC emissions intensity are based on the (California) Air Resources Board and RFS rules and the Laborde (2011a) analysis that is the basis for proposed ILUC emissions limits in Europe.

Some estimates go even higher than 175 gCO_2e/MJ (the highest was 350 gCO_2e/MJ from LEITAP in Edwards, Mulligan, and Marelli 2010), but this graphic does not shown the full magnitude of these very high estimates. This report aims to include as comprehensive a characterization of the range in the literature as possible, but inclusion does not imply that the ICCT considers any particular value to be credible.

Sources: Results taken from Edwards, Mulligan, and Marelli (2010), E4tech (2010), Akhurst, Kalas, and Woods (2011), Searchinger et al. (2008), U.S. EPA (2009; 2010a,b,c; 2012), Fritsche (2010), ARB (2009), Tyner et al. (2010), Hertel et al. (2010a), Taheripour, Tyner, and Wang (2011), Al-Riffai, Dimaranan, and Laborde (2010a), Laborde (2011a), Tipper et al. (2009), Dumortier et al. (2009), Britz and Hertel (2011), Wang et al. (2011; 2012), Dunn et al. (2013), Overmars et al. (2011).

For many people, the wide variation in model results is confusing at first sight, and several groups have used the variability to argue that the whole idea of ILUC contains too much uncertainty to be used in regulation. One reason for the confusion is that it is often unclear what drives the differences and how to figure out which results are more or less credible.

What explains the differences? Should policymakers treat all models as equal? What evidence, if any, are the results based on? What assumptions do they work with? Is there a conflict between carbon savings and other public policy goals—for instance, are there cases where low ILUC emissions are predicted because of reduced food consumption by the poor?

This report aims to elucidate the key factors that affect the expected ILUC emissions due to increased biofuel use. By describing the kinds of shifts in production and consumption that lead ultimately to a prediction of emissions from land use change, the study also identifies some important policy considerations that merit attention.

1.1.1. Why are the land use implications of biofuels important?

Some commentators have contended that biofuels should not be held accountable for emissions that happen at one remove[17]. They point out that, for example, the average U.S. Midwestern corn farmer has no control over land use decisions in the Brazilian Amazon and that holding an industry to account for decisions made by third parties is a novel and arguably unfair approach to regulation. This line of reasoning asserts that the greenhouse gas emissions that occur from land use change are no more the responsibility of biofuels than they are of any other activity that uses land and therefore should not be considered in a life cycle analysis. One variation on this argument is the idea that every indirect land use change is somebody's direct land use change, and the direct land use changes ought to be regulated at source (this reflects the national accounting regime under the Kyoto protocol).

These arguments that biofuels producers should not have indirect land use change held against them are unsatisfactory for two reasons: 1) the producers of biofuels wish to claim environmentally friendly credentials, and 2) markets for biofuels are driven by governmental intervention for their purported societal benefits.

In thinking about the first point, consider on what basis biofuels producers would be able to claim environmentally friendly credentials. In general, especially for 'first-generation' biofuels from crops like corn, wheat, or soy, biofuels are likely to perform worse on most environmental metrics than gasoline[18]—gasoline use is not associated with nitrogen leaching, has less impact on biodiversity than does agriculture, uses less water per megajoule, does not require phosphorus or pesticides, etc. Analyses by Yang et al. (2012) and EMPA (2012) both find that the weighted environmental impact of biofuel use is greater than that of fossil fuel use, even excluding indirect land use change. There

17 "Incorporating indirect effects is part of a second-best policy and we should aspire to first-best policies" (Zilberman, Hochman, and Rajagopal 2010, p. 387); "The avoidance or reduction of an ILUC penalty … is unlikely to be sufficient to encourage feedstock producers to adopt mitigation practices" (Ernst & Young 2011, p. 21); "An indirect effect is, by definition, the application of someone else's direct effect to another product or fuel. Once we start doing that, we are breaking down the very principles we are espousing in cap and trade and polluter pays" (U.S. Congress 2009, p. 4).

18 Yang et al. (2012) note that "E85 does not outperform gasoline when a wide spectrum of impacts is considered. If the impacts are aggregated using weights developed by the National Institute of Standards and Technology (NIST), overall, E85 generates approximately 6% to 108% (23% on average) greater [environmental] impact compared with gasoline."

are some potential air quality gains, but the area in which biofuels producers primarily want to claim benefits is reduced greenhouse gas emissions, so for a biofuel producer to claim to be environmentally friendly, it must be able to make a convincing case that it is reducing greenhouse gas emissions compared with fossil fuels.

The standard carbon accounting assumes that when one barrel of oil equivalent of road fuel is supplied by biofuels, then one less barrel of fossil fuel oil will be burned. In many accounting systems, the carbon emissions saved by this 'avoided combustion' of fossil fuel have been given as a credit to biofuels (e.g., the Kyoto Protocol biomass accounting rules, which are reflected in the European Emissions Trading Scheme). However, when cars burn biofuels instead of gasoline or diesel, they continue to emit about the same quantity of carbon dioxide from their exhaust pipes (as shown in Figure 1.3, most transport fuels have combustion emissions intensity in the range of 70–75 gCO_2e/MJ). There is only a net carbon benefit if there is more plant growth in the world because of biofuel mandates (i.e., if more carbon is absorbed from the atmosphere every year), or if there is a cutback in other emissions (e.g., as a result of lower food consumption and metabolization or a reduction in decomposition of wastes). This increase in global biogenic absorption of CO_2, or reduction in respiration, can be thought of as an offset against the carbon dioxide released by producing and burning the biofuel. It is simple to think of scenarios in which there would be no net increase in sequestration, for instance, if biofuels were produced directly by harvesting wood from old-growth trees or if a lot of biomass were cleared by burning to make way for biofuel crops.

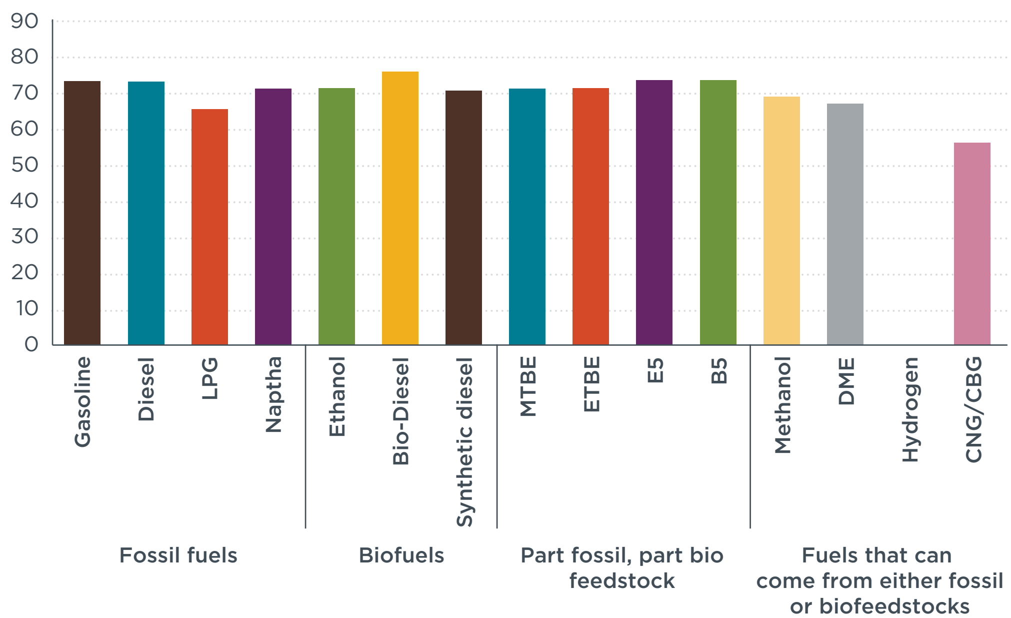

Figure 1.3 Combustion GHG emissions of different road fuels (values for 2010)

Source: Edwards, Larivé, and Beziat, 2011

Just as with any other carbon offset program, in order for biofuels policy to deliver carbon savings, the absorption of carbon must be additional. This additional carbon offset can come in one of four areas:

- Food and feed[19] consumption: If the crops diverted to biofuels are not replaced, there is an offset because those who consume the crops (people and livestock) emit less carbon through their breathing and wastes.
- Average land productivity: If biofuel feedstock is provided through additional growth of crops on existing agricultural land, for example, by boosting yields or by replacing a less productive crop (in metric tons per hectare) with a more productive one, that additional carbon absorption into crops serves as an offset.
- Land use change:
  a. If land is converted to cropland and as a result sequesters more carbon than before in soils and persistent biomass, this yields a carbon credit. More typically, though, expansion into new lands will result in lower carbon stocks and hence will incur a carbon deficit. These emissions have been a focus of the indirect land use change discussion.
  b. In addition to changes to carbon stocks in the period immediately following conversion, there may be a long term change in annual carbon sequestration potential. For instance, growing oil palm trees sequestrate more carbon as biomass annually than grasses would. On the other hand, if annual sequestration is actually less than before (say that annual crops replace a growing forest), then there is a deficit or 'forgone sequestration.'
- Non-land-based emissions changes. This report concentrates only on land use change and related emissions, but there are other areas within and beyond agriculture where offsets could be delivered or additional emissions could occur. For example, the EPA considers emissions changes in methane from livestock and rice paddies in its life cycle analysis for RFS2. Several authors (e.g., Rajagopal, Hochman, and Zilberman 2011) have also noted that biofuels policy could result in changes in the overall consumption of energy. This is sometimes referred to as the 'fossil fuel rebound,' the idea that the reduction in use of fossil fuel energy will be less than the increase in the use of bioenergy.

The net carbon balance from biofuels production over a given period depends on the balance of these positive and negative effects. Land use changes are essential to calculating both the carbon credits and deficits in the equation. In other words, without considering land use change, it is premature to make any statement about whether biofuels reduce global emissions.

If land use change emissions from biofuels were expected to be negligible when compared with combustion emissions,[20] it might be legitimate to ignore them and assign a full credit for all biogenic carbon. However, there is good reason to believe that land use emissions for biofuel production will be significant. For instance, if biofuel feedstock were supplied entirely from conversion of even relatively low-carbon land, significant carbon consequences would still follow. The conversion of pasture in Europe to agricultural use would typically result in a reduction in carbon stocks

---

19 Most agricultural production is for food (71 percent according to FAOSTAT, the statistics division of the Food and Agriculture Organization) or animal feed (26.5 percent according to FAOSTAT); however, there are also industrial uses such as the cosmetics industry's as well as nonfood crops like rubber, cotton, and tobacco. Because food and feed predominate, this report adopts "food and feed" as shorthand, but that could include reductions in crops for these other activities.

20 Looking at land use change due to fossil fuel extraction instead of biofuels, Yeh et al. (2010) find emissions intensity of less than 1 gCO_2e/MJ for California or Alberta oil on average, with a maximum of about 4 gCO_2e/MJ for oil sands mining.

of about 30 metric tons per hectare (owing largely to a reduction in soil carbon storage).[21] This is a much smaller carbon loss than would take place if forestland or other high-carbon ecosystems were converted and might be considered to represent one of the best cases for expansion of farming.[22] Converting that pasture to wheat cultivation for bioethanol would result in ILUC carbon emissions of about 80 gCO_2/MJ, more than from combusting the fuel.[23] This suggests that, unless biofuel expansion can be achieved with limited land conversion, the land use emissions are likely to be on the same order of magnitude as combustion emissions. Certainly, it is impossible to make robust statements about the carbon implications of bioenergy without being able to argue for limited land use change.

A robust assessment of ILUC is therefore a prerequisite for biofuels producers seeking environmentally friendly credentials. For most products, this would not be an existential problem since they would have a market even without environmental benefits. However, in almost all cases worldwide, biofuel production is driven by government intervention. There are several possible reasons to incentivize biofuels, with three being most often quoted: reducing greenhouse gas emissions, supporting rural development, and enhancing energy security. If reducing greenhouse gas emissions is an important policy objective, the governmental authority setting biofuels policy needs to verify whether the policy will indeed deliver greenhouse gas savings. If environmental benefits cannot be demonstrated, there is at best a weak case for the policy.

Unlike some environmental objectives, climate change mitigation is fundamentally global. It makes no difference to the U.S. government whether CO_2 comes from Minnesota or Malaysia; the effects will be the same (whereas shifting U.S. conventional pollution to another country would at least deliver benefits for Americans, even if the other country might suffer). From a policy assessment viewpoint, it is therefore sensible to include known, quantifiable, and significant emissions even when they do not occur on the land from which the biofuel feedstock is actually being sourced. It follows that if a regulator, based on the best available evidence, believes that a carbon mitigation strategy will not reduce global carbon emissions, then that is clearly a bad climate policy. Refusing to incentivize a given biofuel in light of such concerns is not a question of unfairly holding farmers responsible for something they cannot change; it is a question of science-based policy setting.

1.2. INDIRECT LAND USE CHANGE MODELING

As mentioned above, modeling of indirect land use change, and in particular equilibrium economic modeling, has become central to the ILUC debate. This reliance on modeling reflects a desire to quantify the magnitude of indirect land use change emissions. In particular, modeling attempts to answer the question of whether a given bioenergy pathway can be a useful (and cost-effective) climate mitigation strategy.

---

21 Emissions from land use change are discussed more thoroughly in section 3.6.

22 Pasture conversion is one of the better options, but the optimum case for land use change would be rehabilitating a low-carbon degraded landscape so as actually to increase the carbon storage.

23 This calculation is based on 30 metric tons carbon loss per hectare and the EU's Renewable Energy Directive default wheat and ethanol yields.

The most widely quoted modeling attempts use partial[24] or general[25] equilibrium economic models. Using an economic model to estimate ILUC involves the following steps:

1. Choosing a baseline against which to compare the results of the modeling. Predicting the total area that would be under cultivation if various countries pursued ambitious biofuels strategies would be useless in isolation. What is important is the change in land use patterns, compared to a counterfactual world without biofuels. Some models set a baseline in the future—a projection for what the world might be like in, say, 2020 without increases in biofuel demand. These include MIRAGE for the European Commission and FAPRI-FASOM for the EPA. Others set their baseline as current, such as the GTAP modeling for ARB's LCFS. Modelers use the same year for their scenarios as is used in the baseline. For instance, MIRAGE compares a prediction for global land use in 2020 if additional biofuels are cultivated against the counterfactual of extant biofuels policies continuing through that same year. In contrast, GTAP models an immediate increase in biofuel demand and compares it to the present situation. In principle, a baseline could be defined with no biofuels at all. Generally, though, the baseline accommodates some pre-existing level of biofuel use.
2. Defining the scenario. Various modeling exercises aim to assess different policies and answer different questions. U.S. modeling normally aims to examine the effects of U.S. policies like RFS2 or LCFS. GTAP modeling for CARB, for instance, looks at the possible land use consequences of the levels of biofuels adoption expected under the LCFS, compared with existing levels. This involves applying a 'demand shock' to the model—that is to say, having arrived at an equilibrium solution as the baseline, the next step is to increase biofuels demand and have the model come to a new equilibrium. European modeling (e.g., MIRAGE) tends to incorporate the effects of the Renewable Energy Directive. MIRAGE first allows the initial equilibrium to evolve to a 2020 equilibrium (considering external events like population increase that will affect agricultural markets). This generates the baseline. The model is then run again applying a demand shock, forcing biofuel use in Europe to increase each year to 2020 in line with the Renewable Energy Directive targets and letting the model progress to an adjusted 2020 equilibrium that includes elevated biofuel use.
3. Having established a baseline and run a scenario, the modeler must compare the two levels of land use. Presumably, land use is higher in the scenario involving increased biofuels production—the difference must be assessed for each region that is modeled.
4. The change in land use must be split between different land types. For GTAP, this is done endogenously in the model—it includes only categories of managed land and internally distinguishes between pasture, managed forest, and cropland (and, latterly, cropland-pasture). For other models, the type of land converted may not be determined within the model. FAPRI, as an example, calculates the total land use change in a region, and the EPA then estimates how this might be split between forests, grasslands, and such based on historical patterns. MIRAGE falls in the middle: the model determines the split of expansion between man-

24 Partial equilibrium models focus on one specific sector (in this case, agriculture) and do not allow adjustments in other areas of the economy in response to demand changes in the model.

25 General equilibrium models attempt to encapsulate the entire economy. This normally requires sacrificing some of the sectoral detail of the partial equilibrium models but allows more ways to adjust to demand changes.

aged pasture and forest and unmanaged lands, but because the specific land type within the category of unmanaged land is not defined in the model, it uses historical data regarding land expansion to allocate between categories such as unmanaged grassland and unmanaged forest.

5. Once the changes in cultivated area are established, the modelers must assign carbon emissions to them. In general, economic models do not contain assumptions about the carbon stock changes when land uses shift, so this information has to be applied ex post facto. For instance, for the original ARB ILUC factors, the carbon stock changes were calculated based on the Woods Hole Oceanographic Institution dataset.

There are numerous challenges in modeling indirect land use change. The most fundamental and unavoidable challenge is common to any modeling attempt that looks to establish a counterfactual: it is not possible to peer into the future, or to lift the lid off the universe to explore a parallel reality in order to know what would have happened if biofuel mandates were either larger or smaller. In reality, every individual decision regarding whether to convert a given piece of land is based on myriad local circumstances. The best that the modelers can do is to try to reflect the most likely behavior, based on their understanding of how decisions are made and on what has happened in the past. Provided that the many local decisions average out to approximate economically rational behavior, a model can still aim to give a reasonable estimate. This is what Plevin et al. (2010) refer to as 'epistemic uncertainty.'

Another fundamental challenge is building a model that actually provides a good representation of behavior in the real world. Economic models are complicated pieces of software; nevertheless, they are forced to radically simplify complex economic processes. Land use assumptions in GTAP and MIRAGE, for instance, are heavily based on the use of 'constant elasticity of transformation' functions. In these functions, land uses within a given model region are able to switch in accordance with a formulaic approximation of the world. In brief, if the price of one crop increases, then the model would expect a switch from other land uses toward that crop. The magnitude of these changes in land use is determined by a parameter called elasticity of transformation (a higher elasticity means more switching) but also by the initial state: if almost all the land in a region is already cropped, the function will assume less land use change than if there is more of a balance between land uses at the start. Often these functional forms will provide a reasonable approximation of real average behaviors, but there will always be real-world exceptions attributable to more complex political, legal, social, or economic issues. There will always be a limit to how well a model can capture these multifarious motivations.

Other models, however, may use different functional forms, model different regions, and choose different sets of crops. If one crop is particularly important to the results, then it might affect the outcomes significantly if it is not modeled individually but (as with palm oil in GTAP, which is grouped with oilseeds generally) is aggregated with others. The treatment of international trade is also important in the models, especially for investigating effects in countries other than the one that prompted the biofuel demand increase. Some models use a treatment called 'Armington elasticities' that restricts or encourages trade between certain nations. These are normally based on some analysis of historical trading patterns. However, if a new region is likely to emerge as a key feedstock source, this would be difficult to model based on historical trade relations. Other models assume that there is a single world market in which all nations trade equally freely. This approach could fail to capture preferential trading relationships—for instance, the United States trades more with Canada than with Iran, and the model ought to reflect this.

These types of questions, questions about the fundamental appropriateness of the model being used and its ability to predict land use changes are what Plevin et al. (2010) refer to as 'decision uncertainty.' While several models have been tested and subjected to calibration against real data, this is difficult. In many cases the models were originally designed to resolve quite different types of questions, and therefore much of the testing will not have focused on land use change prediction. There is a risk that some technical characteristics of models may lead them to overestimate or underestimate ILUC systematically.

The third major category of uncertainty relates to data. Models are built on a sometimes bewildering array of parameter inputs, notably, data about world agricultural trade and the elasticities that define the relations between pairs of commodities, land types, products, and so on. Agricultural trade data can sometimes be lacking in the areas most relevant to the modeling; for example, the GTAP team at Purdue has had to work extensively to add trade in biofuels to the GTAP database to facilitate its modeling work. Elasticities can be calculated from historical data, if available, but in reality, they often are not. Data may be limited, and often values will only be available for some regions (it is not unusual for trade relations in all regions to be modeled on parameters calculated for the United States). Where elasticities have been published, it is burdensome to test the quality of every calculation. In some cases values in the literature may be based on periods in which a very different policy regime was in place and may therefore be unrepresentative of the current situation. If the relationships between a few critical parameters in a model are off, that could severely alter the outcomes, yet, given the complexity of the modeling, the linkage between a given parameter and the eventual answer may not be clear.

1.2.1. Other methodological questions

This report focuses on effects in the real economy that determine the size of indirect land use changes and on how those effects are represented in the models. There are other modeling choices that may affect outcomes that will not be considered here. One concerns timing. In Europe, regulators have followed the Intergovernmental Panel on Climate Change convention for time accounting of land use change emissions by spreading (or 'amortizing') them over a 20-year period. In America, in contrast, 30 years has been adopted as the amortization period. This means that, for the same set of land use changes, a U.S. model would only report two-thirds of the ILUC emissions reported by an EU model. There are also important questions that revolve around defining how to model the biofuels policy scenario and its counterfactual. One question is whether ILUC factors should be modeled based on a small additional biofuel demand or a large one? There is evidence that the results of some models tend to be approximately linear as total biofuel demand changes, but this may not always necessarily be the case. What happens to food consumption in the baseline scenario—if food consumption shrank, maybe there would be more low-carbon land available for biofuel production? Should ILUC be modeled on a static basis (comparing the boost in biofuel demand and the counterfactual at a single point in time, as done in GTAP) or should the model be dynamic, so that the future can be modeled more explicitly (as in MIRAGE or FAPRI-FASOM)? Does it make a difference if demand for several biofuels is modeled at the same time (should biodiesel demand, to take an example, be included or excluded from the baseline for the corn ethanol scenario)? All of these modeling questions are important but will not be covered in this report (c.f. section 1.3).

1.2.2. Noneconomic modeling approaches

As well as these economic modeling systems, there have been a smaller number of attempts to use simpler, spreadsheet-type models to look at possible ILUC scenarios. Uwe Fritsche at the Öko-Institut (Fritsche 2010) suggests a system based on a 'deterministic' analysis of existing trade flows and carbon implications of continuing historical patterns of cultivation expansion. The institute has determined that the average carbon cost of land expansion for biofuel crops in 2005 was 270 metric tons of CO_2 per hectare and makes a case that if new biofuel feedstocks are sourced at most 75 percent from land expansion, this implies a maximum ILUC emission of 10.2 metric tons of CO_2 per hectare per year (amortized over twenty years). In this approach, all fuels are assumed to affect land distribution in the same way, so the ILUC emissions per hectare are divided by the yield per hectare for each crop to give crop-specific ILUC. While this approach has the benefit of simplicity, it may represent poorly the ILUC risk associated with some crops. Notably, because palm oil has higher yields than any other vegetable oil, it would be assigned the lowest ILUC factor. In reality, there is reason to believe that palm plantation expansion is particularly strongly connected to deforestation and peat destruction (Mietinnen et al. 2012), and the Öko-Institut ILUC factors would fail to capture this.

A second approach comes from the consultancy E4tech, which has built a 'causal descriptive' modeling framework for ILUC (E4tech 2010). In this scheme, the consultants attempt to make predictions based on expert knowledge and analysis of historical patterns about what the main responses to an increased biofuel mandate will be. Unlike an economic model, in which the results characterize the sum of thousands of larger and smaller changes across the economy, a causal descriptive model assumes that a few trading relationships and land use patterns will be dominant. As an example, in E4tech's modeling of ILUC for European oilseed rape biodiesel, it considers changes in the cultivated area of oilseed rape only in Europe itself, the Ukraine, and Canada. In a global economic model, in contrast, changes would be predicted for many other countries. The causal descriptive model also ignores the more intricate economic relations, so effects including the possibility that changes in feed prices might cause an important shift in livestock feeding patterns are not allowed for. The advantage of a causal descriptive framework is the relative transparency about the primary market dynamics that its creators believe will determine the result. The potential disadvantages are that the results may ignore more complex economic interactions that are actually important and that, by leaving so much to expert judgment, it would be possible for different experts to produce wildly different results.

There have also been some papers attempting to use historical statistical analysis to identify ILUC,[26] with inconsistent results. These approaches will not be discussed here. What is of primary interest is the range of possible responses to increased biofuel demand and the land use implications of those responses rather than discussing what has happened to date. Moreover, given the novelty of large-scale biofuel markets and the fact that they are still small relative to other uses of agricultural commodities, it may not be possible to derive statistically useful information about the impact of biofuel demand on land use compared with other market influences (although some useful correlations and apparent causation may be identifiable). Finally, the analyses have been contentious, and a discussion of the relative merits of statistical approaches to this type of historical data analysis is beyond the scope of this paper. Certainly, this type of work does not appear to be mature enough to draw any convincing conclusions from the results.

26 Arima et al. (2011), Kim and Dale (2011), Oladosu et al. (2011), Overmars et al. (2011).

1.2.3. Modeling: In conclusion

With this wide variety of modeling approaches, each based on complex sets of interacting parameters, it can be challenging to get a sense of what is really driving the results. The Joint Research Centre (Edwards, Mulligan, and Marelli 2010) attempted to facilitate discussion of model differences by breaking down the disparate model results in various ways, and several papers have done likewise more or less successfully. This report does not aim to explain why each model gives the results it does or to break apart the esoteric details of various functional forms and economic assumptions. However, the model results provide a useful lens through which to consider ILUC. The results from each model can be understood as mapping out a scenario for the way that biofuel demand expansion will happen in the real world, and exploring what the models are predicting[27] can help to illuminate the discussion of what would contribute to either larger or smaller indirect emissions. The following chapter uses the model results as a vehicle to explore different answers to questions like, "Will food consumption drop, and where?"; "What will happen to yields?"; and "Will agricultural expansion encroach on forests or pastures?"

1.3. THIS REPORT—A GUIDE FOR THE PERPLEXED

> *The generosity of the Earth allows us to feed all mankind; we know enough about ecology to keep the Earth a healthy place; there is enough room on the Earth, and there are enough materials, so that everybody can have adequate shelter; we are quite competent enough to produce sufficient supplies of necessities so that no one need live in misery.*
>
> —E .F. Schumacher, from *A Guide for the Perplexed* (Harper Publishing, 1977)

> *We naturally like what we have been accustomed to, and are attracted towards it.... The same is the case with those opinions of man to which he has been accustomed from his youth; he likes them, defends them, and shuns the opposite views.*
>
> —Moses Maimonides, from *The Guide for the Perplexed* (12th century A.D.)

> *It turns out that despite the complexity of the models, only a handful of easily understandable assumptions are important in determining the simulation results. By showing the effect of these assumptions on the predicted economic costs, not just in one particular model but in all of them, this report can help readers to apply their own judgments about which models are more realistic and to reach their own conclusions about which economic predictions are more credible.*
>
> —Robert Repetto and Duncan Austin, from *The Costs of Climate Protection: A Guide for the Perplexed* (World Resources Institute, June 1997)

This study is not the first, and doubtless will not be the last, to review the subject of indirect land use change. The British government's Gallagher Review (UK RFA 2008) assessed six questions about the likely impacts of biofuel demand, with the aim of drawing conclusions about the nature and level of the risks and making recommendations on how these risk should be dealt with in government policy and carbon accounting. The European Commission's Joint Research Centre (Edwards, Mulligan, and Marelli 2010) approached the question by setting up a framework to compare the results of various economic modeling exercises. The European Commission Directorate-General for Energy (EC DG Energy 2010) produced

27 Some (if not all) economic modelers might take issue with the use of the term "prediction" to describe their work. It must be understood that the real world is subject to many more and often larger economic stimuli than just biofuel mandates, and so a model prediction for 2020 should not be understood as a prediction of what the world will actually be like in 2020 but a scenario to allow for comparison. The prediction is not of how the world will be, but of the difference that added biofuel demand is likely to make.

a literature review focused on comparing "methodological and data choices" in various studies attempting to quantify indirect land use change. The U.S. Department of Agriculture reported to Congress (Marshall et al. 2011) on the state of knowledge about the drivers of land use change and the models used to assess it, with the intention of providing a neutral survey of the literature. Ecofys, a renewable energy consultancy, reported to the Global Bioenergy Partnership (Ecofys 2011) as part of its discussion of the indirect effects of biofuels production, with a focus on explaining modeling attempts to quantify indirect effects and exploring options to avoid or minimize them. California's Air Resources Board Expert Workgroup on Indirect Land Use Change (2010) produced a series of reports considering in detail a range of questions critical for California's modeling of ILUC. There are other papers and reports available covering the same areas to a greater or lesser extent.

This report does not aim to be a comprehensive literature review in the style of the DG Energy or U.S. Department of Agriculture reports. It is not built around a new quantitative analysis in the way that the JRC model comparison is. Rather, as the World Resources Institute (Repetto and Austin 1997) did for the cost of climate action in *The Costs of Climate Protection: A Guide for the Perplexed,* this report aims to identify and explore the key factors that determine the amount of indirect land use change occurring in the real world. Based on this analysis, it looks at how these factors are represented in the major models being used for regulatory purposes and how they influence the estimates on which policymakers must base their decisions.

Given the many hundreds and thousands of pages devoted to this topic, it is not a simple thing to make a truly useful addition to the corpus. This report tries to take a step back from the specifics of this or that economic model, in order to provide a detailed but accessible walk-through of the six principal determinants of indirect land use change emissions, not only in the models but also in the real world. Throughout the report, the modeling exercises are brought back in an effort to illuminate how the six determinants have been dealt with and how that has affected the results. With its scope restricted to the science of ILUC, the report does not consider other indirect emissions implications of bioenergy, nor does it attempt any comparative analysis of the different policy approaches that may be available to deal with ILUC. Discussion about ILUC models here is largely restricted to those associated with regulatory action in the United States and proposed regulatory action in the EU. There is no attempt to make an overall recommendation about which model is better or worse, rather to provide a framework to allow the reader—whether an academic, a regulator, a policy official, an investor, or just an interested member of the public—to make an informed decision about the plausibility of the results of different models, notably, whether it seems reasonable to believe that a given biofuel pathway will actually deliver a net carbon emissions reduction compared with the fossil fuel production it seeks to replace.

In the end, having read this report the reader should be better placed to understand what the results of a given model really mean, and most importantly to be able to make a better informed judgment about whether it seems reasonable to believe that a given biofuel pathway will actually deliver a net carbon emissions reduction compared to the fossil fuel it aims to replace.

2. DIRECT EMISSIONS

Before examining indirect land use change, it is worthwhile to take a brief look at the direct emissions associated with fossil fuels and biofuels. 'Direct emissions' means the greenhouse gas (GHG) emissions that result from both fuel combustion and the production processes to make the fuels. It has already been shown that combustion emissions intensity is around 70–75 grams of carbon dioxide equivalent per megajoule (gCO_2e/MJ) for all of the main liquid transport fuels (Figure 1.3). But the emissions resulting from production can show wide variation depending on feedstocks in particular but also on extraction energy (for fossil fuels), agricultural practices (for biofuels), and processing efficiency.

Figure 2.1 shows the well-to-wheels emissions intensity of fossil fuels, as modeled by Jacobs (2009). There is a relatively limited overall variation in total well-to-wheels emissions intensity of about 20 gCO_2e/MJ. This variation is largely driven by the amount of energy used in oil production (oil sands especially require large energy inputs to produce), the amount of energy required to refine different specifications of oil, and the volumes of associated gas that are flared or vented.

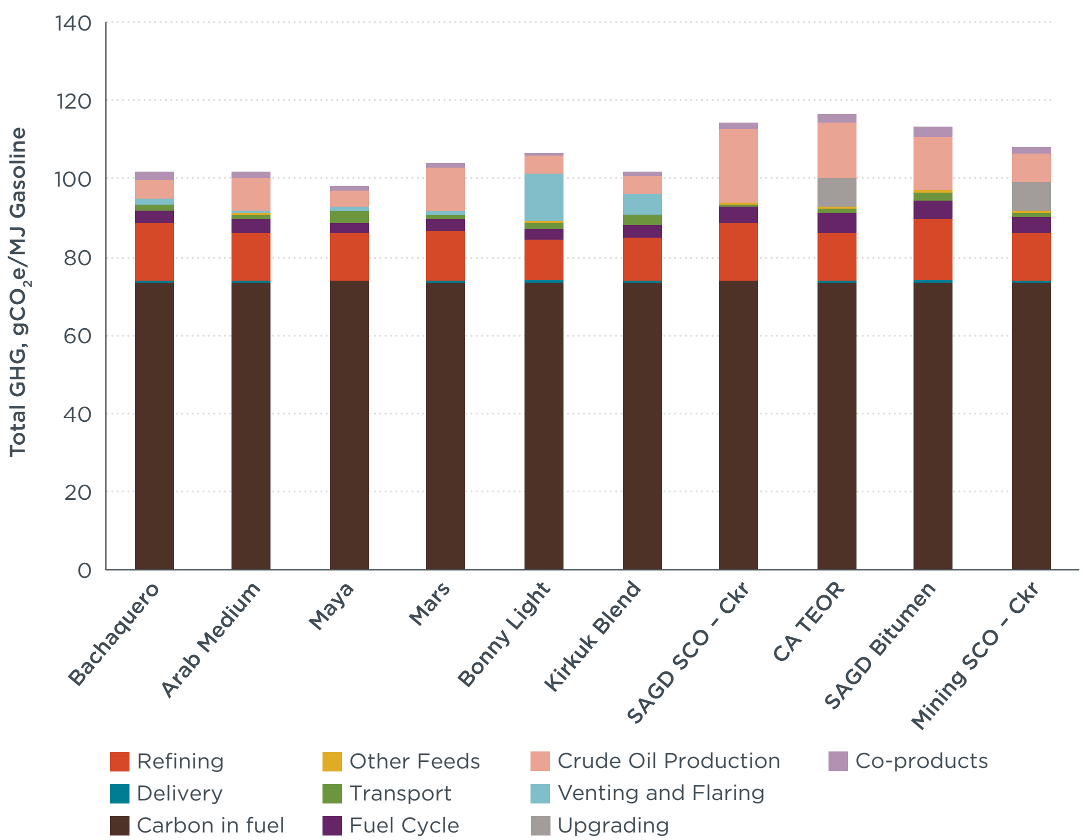

Figure 2.1. Variation in WTW GHG emissions for crude and bitumen

Source: Jacobs, 2009

Figure 2.2 shows the variation in the carbon intensity of biofuels supplied in the United Kingdom in 2009/2010. For biofuels, the difference between the highest and the lowest emissions intensity is much higher than for fossil fuels, more than 100 gCO_2e/MJ.

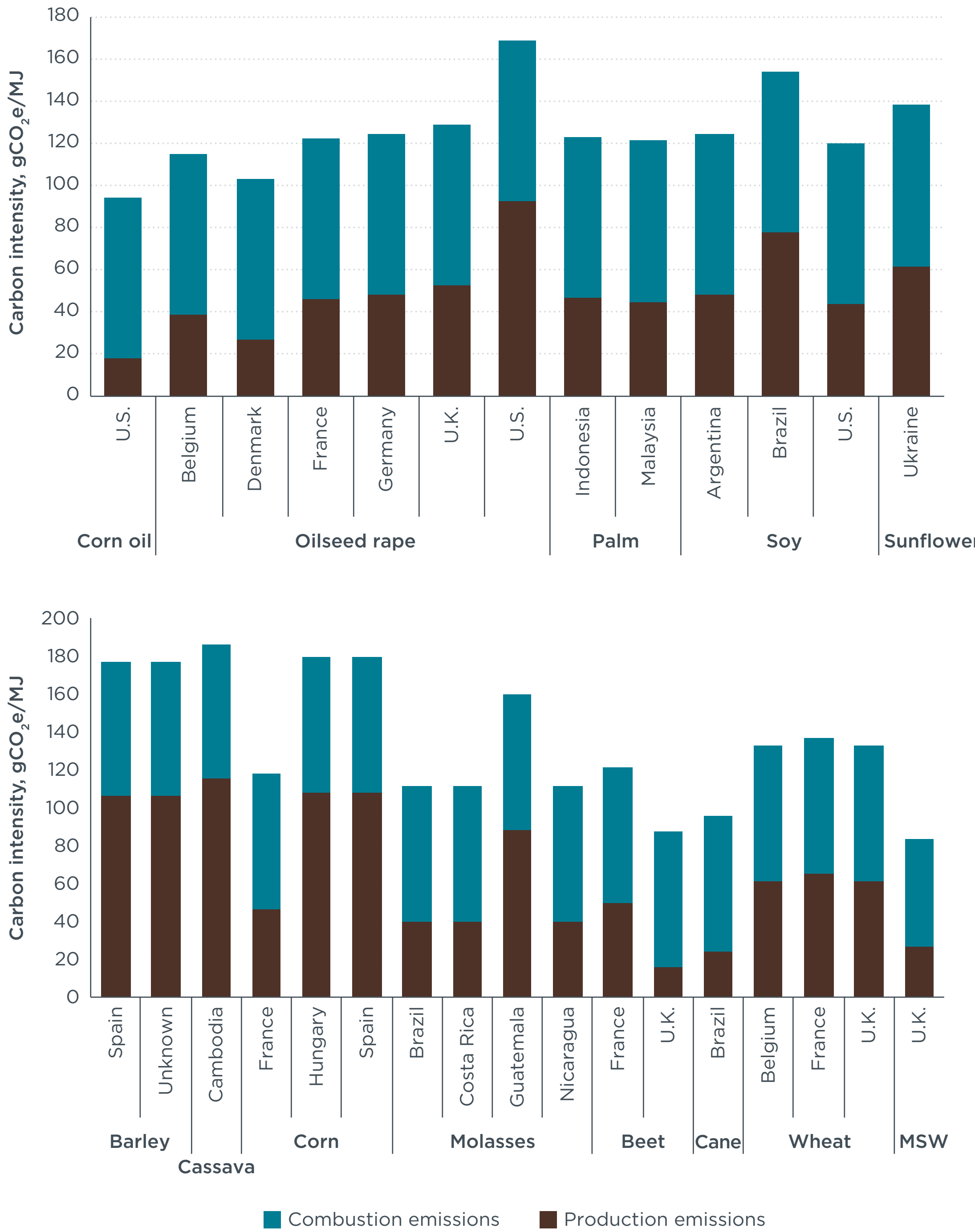

Figure 2.2. Variation in carbon emissions for selected biofuel feedstocks and countries of origin as reported to the UK Renewable Fuels Agency for 2009/10: biodiesel (top) and ethanol (bottom)

Source: Renewable Fuels Agency, 2011

MSW = municipal solid waste

Note: It is conventional in biofuel LCA to give a credit to biofuels equivalent to their combustion emissions. Combustion emissions are included to be consistent with the argument that a robust ILUC analysis is necessary before a biogenic carbon credit should be given.

Among the production emissions, the contributions made by the cultivation, processing, and transport components can also vary widely. Figure 2.3 shows the emissions differences between various feedstocks as defined in the EU's Renewable Energy Directive.

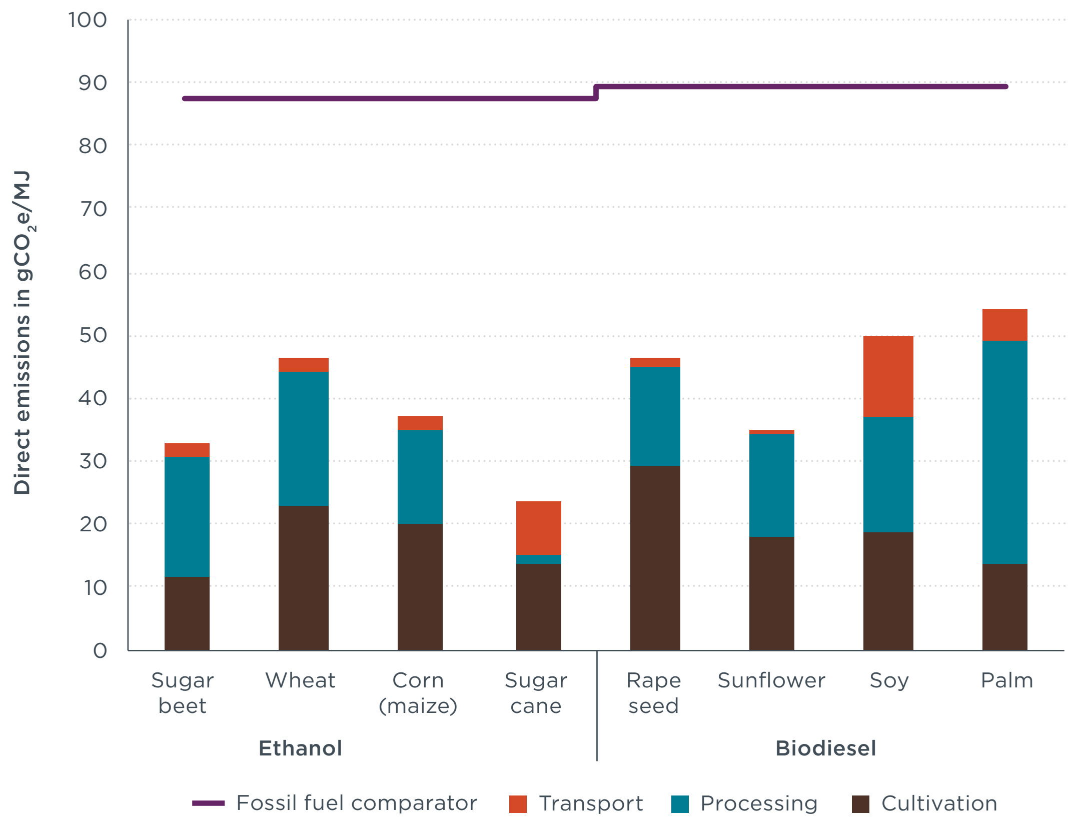

Figure 2.3 Typical production emissions intensity as defined in the European Renewable Energy Directive, split into transport, processing and cultivation

Along with variation between fossil fuel feedstocks and between biofuel feedstocks, there is a degree of variability in quantifying the emissions intensity even for a particular feedstock. For instance, the typical emissions for palm oil biodiesel quoted in the Renewable Energy Directive range from 32 to 54 gCO_2e/MJ, depending on whether methane is captured.

Beyond variation stemming from different production practices, there can be genuine uncertainty in the estimation. Issues in life cycle analysis such as the rate of nitrous oxide release due to fertilizer application are still being researched and disputed (e.g., Reay et al. 2012). Nevertheless, the challenge of improving the accuracy and certainty of direct emissions estimates is relatively well defined, and it is known which additional research would allow direct emissions to be increasingly tightly quantified. With indirect land use change, by contrast, it can often seem unclear how one would go about improving estimates since ILUC cannot be neatly identified in historical data or measured with instruments.

3. WHAT ARE THE DETERMINANTS OF ILUC EMISSIONS?

It is a simple tenet of economics that an increase in demand for a product, all other things being equal, will manifest itself in an increase in price for that product. This premise is central to the concept and the economic modeling of indirect land use change (ILUC).

In an economic sense, indirect land use change occurs because when biofuel markets expand the return to investment from bringing new land into production improves. Hence, investors will convert more land to cultivation in a world with active biofuels policies than otherwise. However, as outlined in Chapter 1, land expansion is only one of several possible responses to higher food commodity prices. To get an appropriate estimate of the likely scale of emissions from indirect land use changes, one must also consider the other responses to increased biofuel mandates.

Informed by existing economic modeling and other studies in the field, this report identifies six primary factors that determine the size of the emissions expected from indirect land use change. These dictate how much of the response to stricter biofuel mandates will happen on the demand side, how much on the supply side (and how much of that from land expansion), and what the carbon implications of those land use changes are.

The first factor, determining whether people eat less food as prices change, is ***elasticity*[28] *of food demand to price.*** If food consumption is highly elastic relative to price, it means that a small price increase spurred by biofuel demand would cause people to eat much less, making the uneaten material available for conversion to biofuel. On the contrary, if food demand is relatively inelastic to price, then even a large price change would have little effect on diets, and the material for biofuels production would need to come from somewhere on the supply side.

Working out how the productivity of the agricultural system changes as a result of new crop demand is more complicated. Three parameters interact to determine the overall productivity effect (and each of these could in theory be broken down even further). The first of these is the ***elasticity of yield to price***. This factor determines how the average yield of a given crop changes in response to a price change. If raising yields is easy, and farmers are very responsive to price, yield increase can deliver the raw material to make biofuels without requiring large areas of new land or forcing people to eat less. If, on the other hand, raising yields is difficult or farmers are unresponsive to prices, then each field will still produce the same amount of crops as before, and the biofuel feedstock will need to come from new land or reduced consumption.

The second determinant of productivity is ***choice of crops.***[29] Some crops are more productive, in terms of metric tons of raw material produced per hectare, than others. The nutritional content may vary, but if the focus is on sheer quantity of production then growing more of the highest-yielding crops, and less of crops with low yields, can change the productivity of the system as a whole.

---

28 In economics, the elasticity of some quantity (such as demand or yield) to price is a measure of how much that quantity changes if prices change.

29 This can include choosing whether to raise cattle on pasture, which has a lower productivity in terms of food yield per hectare, or to grow crops for direct human consumption, which is much more productive.

The third factor is ***utilization of co-products.***[30] Consider corn: from which only about two-thirds of the material in the grain can be fermented into ethanol. The rest, including proteins and fats, is left over for other uses or disposal. The least productive fate for this co-product would be simple discarding; an alternative use might be to burn it for heat and power, but the standard current usage is as animal feed. This use is also the most efficient in terms of limiting ILUC, as it reduces demand for other grains. If co-products are well utilized, that is equivalent to increasing the productivity of the land.

If productivity gains are not adequate to meet the need for increased feedstock supply, the rest must come from growing crops on new land. The rate at which land expansion occurs is determined by the ***elasticity of area to price.*** If area is greatly price elastic, it means that farmers will respond to a small price change by clearing and cultivating lots of new land, so that there will be little need to increase yields or eat less, and the ILUC will be large. In contrast, if area elasticity is small compared to food consumption and yield elasticity, then only a small area will be converted.

The final factor affecting the magnitude of ILUC emissions is the ***carbon stock of new land.*** The balance of productivity increase, demand reduction, and land expansion dictates how many new hectares are needed, but translating that into an ILUC emissions intensity requires knowing (or estimating) how much carbon is released when that land area is cleared. If new land is taken primarily by clearing forest, the emissions are likely to be high. If new land can be found with sparse vegetation and low levels of organic carbon in the soil, the emissions will be much less.

The following sections explore each determinant in turn, looking at what can be learned from the literature and how they have been captured in previous economic modeling.

3.1. ELASTICITY OF FOOD DEMAND TO PRICE

The initial market response to increased demand for biofuel feedstocks will be to raise the price of those feedstocks. It is simple economics that increasing the price of some good will tend to reduce its level of consumption. If the price of cigarettes goes up, fewer people will smoke. If the price of televisions goes up, people buy fewer new televisions. In the same way, if the price of feedstocks rises due to increased demand for biofuel, we expect to see consumption of feedstocks fall in other sectors of the economy. The current generation of biofuels is based on crops that would otherwise largely supply food and feed.[31] 'Food' in this context means grain or produce intended for human consumption, such as wheat to make bread. 'Feed' (or 'fodder') refers to crops destined for animal consumption, such as wheat being fed to pigs. When increased biofuel demand raises the price of agricultural commodities, it is therefore expected that a reduction in food and feed demand will follow.

30 Co-products are different products derived from the same feedstock. For instance, in the case of oilseeds, one co-product from oil pressing is vegetable oil (which can be used for biodiesel), while a second is oil meal (which can be fed to livestock).

31 There is also some level of competition with other markets for clothing fibers, cosmetics, and so on. For palm oil, for instance, cosmetics markets are currently of an importance comparable to or greater than biofuels markets—hence the sharper focus in the past by campaigning organizations such as Greenpeace on deforestation driven by cosmetic brands such as Unilever's Dove (see, e.g., http://www.greenpeace.org/international/en/multimedia/photos/stop-dove-destroying-forests-f/) than on palm oil as a biodiesel feedstock. Rather than constantly saying "food, feed, clothing fiber, cosmetic and other demand", subsequent instances will simply reference "food and feed demand" as a shorthand, as these remain the most significant markets (according to the Food and Agriculture Organization's statistics division, FAOSTAT, 71 percent of agricultural output is for human consumption and 26.5 percent for animal feed). This does not mean that those other uses of agricultural resources are not subject to the same price dynamics—they are. There can also be knock-on consumption impacts on commodities like cotton that are not themselves biofuel feedstocks but that compete with feedstocks such as corn for land.

When modeling indirect land use change, the projected magnitude of this drop in consumption is one of the key results. If a price rise from adding 100 million metric tons of corn demand for biofuels results in other sectors using 50 million metric tons less, this substantially reduces the net increase in demand for corn and hence any need for land expansion. If, on the other hand, adding 100 million metric tons of demand for biofuels reduces demand in other sectors by only 5 million metric tons, this will have only a marginal effect on net land demand. Thus, the larger the effect of biofuel mandates on food consumption, the less indirect land use change is expected. It is important in making policy decisions about indirect land use change that this dimension is considered. In particular, it might be considered undesirable to incentivize a biofuel feedstock specifically because a model predicts reduced consumption of that feedstock as food in the developing world, but this would be one possible outcome of using ILUC factors without consideration of food effects.

Roberts and Schlenker (2010), whose results and methodology shall be discussed further in section 3.5.1, use historical econometric analysis to compare the size of the demand response to price (reduced food consumption) with the supply response to price (more production through increased area under cultivation and productivity gains) for the world's four major crops: wheat, corn, soy, and rice. They find that there is a statistically significant demand response to price and that it is somewhere between half and the same size as the supply response. This suggests that (assuming the four major commodity crops are representative of the system as a whole), one might expect reductions in consumption to free up between a third and a half of the feedstock required to meet biofuel mandates.

Increases in staple food prices are expected to have more deleterious welfare effects for poorer people than for richer people, and in the developing world than in the developed world. In general, it is assumed that demand elasticity will be greater among the poor, for several reasons. First, the poorer a person is, the higher the percentage of household income that must be spent on food staples.[32] For a poor African, food purchases will represent a larger percentage of overall expenditures than for a rich American. Second, the prices people in the developing world pay for food tend to be more sensitive to underlying commodity prices because they are eating staple food with less value added.[33] To give a simple example, a 50 percent increase in corn prices could mean a 25 percent increase in the overall food bill for a poor African but perhaps only a 1 percent increase (or even less) in the food shopping bill for a rich American. The combination of these effects might mean that, for a poor African, a 50 percent price hike for corn would represent a very noticeable change, and it might be difficult to maintain expenditure on food in response. For a rich American, on the other hand, the increase in grain prices would represent a small increase in the food bill, one that could be easily afforded and, indeed, for many people, hardly noticed. Given this line of reasoning, it would seem logical that in the event corn prices did increase 50 percent, the demand response among the African poor would be much greater than among the American rich.

While this narrative seems plausible, it is possible to identify circumstances that might increase developed world demand elasticity as well. It was noted above that food for people in the developed world is often more heavily processed than in the developing world; that

32 International Comparison Project (ICP) 1996 data suggest that the average consumer in a low-income country spends more than 50 percent of income on food, beverages, and tobacco (Seale, Regmi, and Bernstein 2003).

33 Seale, Regmi, and Bernstein (2003).

is, the price of food is more reflective of labor and other costs than the costs of the raw food commodities used to manufacture it. While this insulates consumers from commodity prices, the company doing the food processing may pay a lot more attention to prices than the consumers themselves would, and look for opportunities to reduce raw material inputs. As a simple example, it was widely reported in the British media in late 2011 that the chocolate company Cadbury was reducing the number of chocolates in each box without lowering the price.[34] The company explicitly blamed the change on the rising cost of cocoa. While it seems likely that some consumers will respond by simply buying more boxes of chocolate (inelastic demand for cocoa), other consumers may well end up eating less, perhaps without realizing the difference (elastic demand for cocoa). Reduction in the consumption of food commodities by intermediate producers (like chocolate companies) is an important consideration in the International Food Policy Research Institute's Modeling International Relationships in Applied General Equilibrium (IFPRI-MIRAGE) modeling for the European Commission by David Laborde (2011a).[35]

The Economic Research Service (ERS) of the U.S. Department of Agriculture (USDA) reported in 2003 on the question of how food consumption response to price varies with income. The report, using 1996 data from the International Comparison Project (ICP), finds clear evidence that "demand responses to price changes are also generally the largest for poorer countries and decrease with affluence" (Seale, Regmi, and Bernstein 2003, p. iii). For instance, the price elasticity of consumption of bread and cereals for Tanzania is given as 0.62, while for Germany it is 0.15 and for America, 0.05. Several models reflect this hierarchy of higher consumption elasticity in poorer countries. Hertel et al. (2010b, p. 11) observe, "Direct consumption of coarse grains [grains other than wheat or rice] is only modestly affected in the U.S. (-0.9%), owing to price-inelastic demand," while the International Food Policy Research Institute (IFPRI) draws food demand elasticities for MIRAGE directly from the USDA ERS's work. The ERS also notes that demand for staple foods is less elastic than demand for higher-value products such as meat and fish. This dynamic is also reflected in the Global Trade Analysis Project (GTAP) model. Based on the USDA research, it is reasonable to expect that increases in feedstock prices will disproportionately harm less affluent people in lower-income nations (the impacts should also be greater on less affluent people in high-income nations than on the wealthy).

While there is good reason to believe that for the same change in price, poor people will react more strongly than rich people, a price shock originating from the United States will not be perfectly transmitted to developing world markets. For example, an increase in biofuel demand that raised U.S. corn prices by 20 percent might only lift world prices by 10 percent, and this might result in a 5 percent rise in a certain developing country. This means that people in the developing world are likely to experience a smaller price change than people in the richer countries where biofuel mandates are being put in place. Some countries in particular have food markets that are relatively insulated from global prices, whether by government policy or physical remoteness (the larger the contribution of transport cost to food prices, the more weakly changes in fundamental

34 See http://www.telegraph.co.uk/foodanddrink/foodanddrinknews/8739531/Cadbury-cuts-the-size-of-its-chocolate-tins.html; http://www.dailymail.co.uk/news/article-2033526/Cadbury-s-start-pruning-Roses-First-sliced-chunks-bars-Dairy-Milk-Now-11-sweets-vanish-tin.html; http://conversation.which.co.uk/consumer-rights/food-prices-budget-supermarket-shrinking-products/; http://www.mirror.co.uk/news/top-stories/2011/09/04/cadbury-roses-and-heroes-shrunk-in-size-but-not-in-price-115875-23394541/

35 Laborde notes that intermediate consumption can be reduced because, for instance, of the "decrease of the average contents of flour in processed food" (p. 82).

commodity prices will be felt). For crops like corn and wheat, the most direct impact is also likely to be on feed prices for livestock, and in the developing world poor people are much less likely to be consumers of meat. None of this eliminates the potential for biofuels to increase food insecurity, but it does mean that poor people in the developing world are unlikely to experience the bulk of the change in food consumption patterns. For instance, in the GTAP scenario described in section 4.3, the predicted change in food consumption in Africa contributes only a sixth as much ILUC avoidance as the change in food consumption in the United States.

3.1.1. The 2008 food price spikes and the welfare implications of elastic food and feed demand

Reducing food and feed demand is good for lowering carbon emissions, but it might have other social consequences that would be a legitimate cause for concern. In the public discourse, these concerns have been expressed in what is referred to as the 'food versus fuel' debate.

The public debate has tended to revolve around not the economic modeling results but the idea that competition between the rich for fuel and the poor for food is necessarily a problem, and around economic analysis of the food price spike in 2008 and a similar price spike in 2011. The campaign group ActionAid[36] writes, "In 2008, global food prices rose dramatically causing a world food crisis that led to riots in more than 30 countries. Many experts, including some at the World Bank, cited industrial biofuels as one of the main causes. ActionAid estimates that an extra 30 million more people were pushed into hunger as a result of biofuels during this crisis."[37]

On the other side of the argument, some voices from the biofuels industry have argued that there is in fact no conflict between food and fuel supplies. Tom Buis of ethanol lobbyist Growth Energy,[38] also citing the World Bank, said, "I applaud the World Bank for admitting the error of their ways and setting the record straight. They have dispelled the myths and lies perpetuated by those who tried to say there was a 'food-versus-fuel' issue. This study clearly shows that the notion of food-versus-fuel was simply wrong. In fact, this study confirms what we've known for some time—the impact of ethanol production on food prices is minimal and other factors, including increased oil prices, were the main drivers in the rise of food prices."[39]

The fact of the matter is that there is little question that biofuels have caused and will cause food prices to rise, a conclusion grounded in both study of the 2008 food price spike and analysis of the medium-term price impacts of biofuels using the economic models. Part of the confusion about the World Bank's line, to which Buis was reacting, comes from the publication of two separate papers by different teams. The first paper, from Donald Mitchell (July 2008), says that, "The combination of higher energy prices and related increases in fertilizer prices and transport costs, and dollar weakness caused food prices to rise by about 35–40 percentage points from January 2002 until June 2008. These factors explain 25–30 percent of the total price increase, and most of the

36 ActionAid characterizes itself as "an international organization, working with over 25 million people in more than 40 countries for a world free from poverty and injustice." http://www.actionaid.org/

37 http://www.actionaid.org.uk/doc_lib/biofuels_campaign__guide.pdf

38 Growth Energy characterizes itself as representing "the producers and supporters of ethanol who feed the world and fuel America in ways that achieve energy independence, improve economic well-being and create a healthier environment for all Americans now." http://www.growthenergy.org/

39 http://www.growthenergy.org/news-media/press-releases/world-bank-study-debunks-food-vs-fuel-myth/

remaining 70–75 percent increase in food commodities prices was due to biofuels and the related consequences of low grain stocks, large land use shifts, speculative activity and export bans."

Evidently, the July 2008 World Bank paper puts the onus for the 2008 food price rises largely (70–75 percent) on biofuels. However, it does this by assuming that growth in biofuels markets can be held responsible for speculative activity and export bans. This study goes on to note, "It is difficult, if not impossible, to compare these estimates with estimates from other studies because of different methodologies, widely different time periods considered, different prices compared, and different food products examined, however most other studies have also recognized biofuels production as a major factor driving food prices."

A later study by a different team, which caused Buis to decry "the myths and lies perpetuated by those who tried to say there was a 'food-versus-fuel' issue," represents the application of one of these "different methodologies." The latter paper (Baffes and Haniotis 2010) argues, "that the effect of biofuels on food prices has not been as large as originally thought, but that the use of commodities by financial investors (the so-called 'financialization of commodities') may have been partly responsible for the 2007/08 spike."

Note that much of the difference in conclusions regarding the 2008 food price spike depends on whether biofuels are seen as having driven commodities speculation. Mitchell does view it that way, while Baffes and Haniotis do not. Baffes and Haniotis conclude, "We conjecture that index fund activity (one type of 'speculative' activity among the many that the literature refers to) played a key role during the 2008 price spike. Biofuels played some role too, but much less than initially thought."

That is, while they consider speculation to be an independent and dominant cause, they still believe that biofuels played a significant role. Wiggins, Keats, and Compton (2010) echo this conclusion, suggesting that the 2008 food price spikes were caused by a confluence of medium- and short-term effects (a combination of events that put upward pressure on prices and others that increased potential volatility), with "panic reactions" by governments and investors exacerbating the peak prices reached in mid-2008. They point to the diversion of corn to ethanol as a key short-term driver of the spike (see Figure 3.1 for a depiction of this buildup of medium-, short-, and very short-term impacts). Wiggins and colleagues declare that "expanded biofuels played a substantial role in pushing up maize [corn] prices" but also that "it is unlikely to have been the main factor." This seems a reasonable assessment.

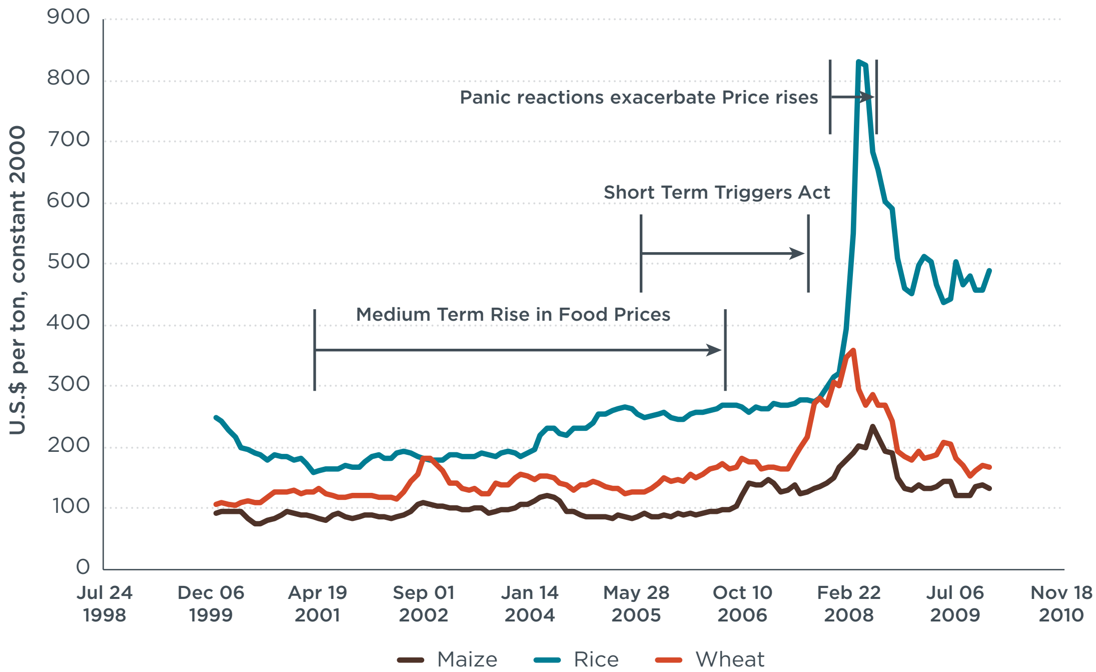

Figure 3.1 Distinction between medium-term, short-term, and very short-term timescales for price increases

Source: Wiggins et al., 2010

Note: Based on International Monetary Fund (IMF) monthly commodity prices deflated by the US GDP deflator

The takeaway from all this is that, while the extent to which biofuels may be a driver of price spikes and volatility is controversial, there is wide consensus, notwithstanding the protestations of Growth Energy and others, that biofuels do increase price volatility and that increased food prices will be a medium-term consequence of biofuel mandates. and Haniotis provide an extensive listing of studies that support this conclusion:

> *"The contribution of biofuels to the recent price boom, and especially the price spike of 2007/08, has been hotly debated. Mitchell (2008) argued that biofuel production from grains and oilseeds in the US and the EU was the most important factor behind the food price increase between 2002 and 2008, accounting, perhaps, for as much as two thirds of the price increase. Gilbert (2010), on the other hand, found little direct evidence that demand for grains and oilseeds as biofuel feedstocks was a cause of the price spike.*
>
> *FAO (2008) compared a baseline scenario, which assumes that biofuel production will double by 2018, to an assumption that biofuel production will remain at its 2007 levels; it concluded that in the latter case grain prices would be 12 percent lower, wheat prices 7 percent lower, and vegetable oil prices 15 percent lower than in the baseline scenario. OECD (2008) arrived at similar conclusions for vegetable oils, finding that their prices would be 16 percent lower than the baseline if biofuel support policies were abolished; eliminating biofuel subsidies would have smaller impacts on the prices of coarse grains (-7 percent) and wheat (-5 percent). Rosegrant (2008), who simulated market developments between 2000 and 2007 (excluding the surge in biofuel production), concluded that biofuel growth accounted for 30*

> *percent of the food price increases seen in that period, with the contribution varying from 39 percent for maize to 21 percent for rice. Looking ahead, Rosegrant found that if biofuel production were to remain at its 2007 levels, rather than reaching its mandated level, maize prices would be lower by 14 percent in 2015 and by 6 percent in 2020.*
>
> *Banse, van Meijl, Tabeau, & Woltjer (2008) compared the impact of the EU's current mandate to (i) a no-mandate scenario and (ii) a mandate whereby the US, Japan, Brazil also adopt targets for biofuel consumption. They estimate that by 2020, in the baseline scenario (no mandate), cereal and oilseed prices will have decreased by 12 and 7 percent, respectively. In the EU-only scenario, the comparable changes are -7 percent for cereal and +2 percent for oilseeds. By contrast, under the 'global' scenario (adding biofuel targets in US, Japan, and Brazil) oilseed prices will have risen by 19 percent and cereal prices by about 5 percent. The European Commission's own assessment of the long-term (2020) impacts of the 10 percent target for biofuels (i.e. that renewable energy for transport, including biofuels, will supply 10 percent of all EU fuel consumption by 2020) predicts fairly minor impacts from ethanol production, which would raise cereals prices 3-6 percent by 2020, but larger impacts from biodiesel production on oilseed prices; the greatest projected impact is on sunflower (+15 percent), whose global production potential is quite limited. Taheripour, Hertel, Tyner, Beckman, & Birur (2008) simulate the biofuel economy during 2001-06. By isolating the economic impact of biofuel drivers (such as the crude oil price and the US and EU biofuel subsidies) from other factors at a global scale, they estimate the impact of these factors on coarse grain prices in the US, EU, and Brazil at 14 percent, 16 percent, and 9.6 percent, respectively."*[40]

Similarly, almost all economic modeling of the future impacts of biofuels on food markets predicts a food consumption reduction resulting from biofuels.

While the preponderance of evidence indicates that biofuel policies will drive up food prices and that this in turn will reduce consumption to some extent, it is less clear by exactly how much prices will change and what impact this will have on welfare. Here, one must be careful in interpreting the conclusions of economic modeling. When talking about food price spikes such as in 2008, it is natural to associate high food prices with severe welfare impacts. In looking at economic modeling results, however, inelastic food demand (a situation in which food consumption **does not diminish**) would lead to higher forecast prices. Thus, in the realm of an economic model, a larger price change could be consistent with predicting a more limited impact on nutrition. Of course, a model prediction that households will spend an increasing percentage of their income on food despite rising prices could still be considered a problem, as this would mean reduced income availability for other goods, potentially including important basic provisions like healthcare and education.

40 Full text available at: http://www-wds.worldbank.org/external/default/WDSContentServer/IW3P/IB/2010/07/21/000158349_20100721110120/Rendered/PDF/WPS5371.pdf

As an example of the difficulty of characterizing the severity of the welfare impacts of biofuels-induced food price changes, consider the credibility of the claim by ActionAid that "an extra 30 million more people were pushed into hunger" by biofuels in 2008. This is a tricky calculation to attempt for a given price spike like 2008 because, as noted, there remains no consensus on the marginal level of the price effect that could fairly be attributed to biofuels (10 percent or 75 percent would both be significant but clearly suggest extremely different scales of welfare impacts).

From Wolfram MathWorld:

The Gini coefficient (or Gini ratio) G is a summary statistic of the *Lorenz curve* and a measure of inequality in a population. The Gini coefficient is most easily calculated from unordered size data as the "relative mean difference," i.e., the mean of the difference between every possible pair of individuals, divided by the mean size μ,

$$G = \frac{\sum_{i=1} \sum_{j=1} |x_i - x_j|}{2\, n^2 \mu}$$

It can be simpler to attempt to model the medium-term impacts of biofuel mandates going forward rather than to try to unpick historical events. Both De Hoyos and Medvedev (2009) and Cororaton, Timilsina and Mevel (2010) try to address this question using a combination of World Bank general equilibrium modeling called ENVISAGE and income distribution modeling using a second World Bank model. Cororaton et al. compare a 'business as usual' 2020 baseline to two scenarios: one with the biofuel targets that had already been announced by various governments and one extended to include more ambitious targets. They find a global increase in absolute poverty (people living on below $1.25 per day), compared against the baseline of 6 million in the "announced" scenario and 7 million in the "extended" scenario. They also find a slight rise in the global GINI coefficient, which is a measure of income inequality (see the text box), so they expect the world to become a little less equitable owing to biofuel mandates. These studies model not only welfare losses as food prices increase but also economic gains from expanding biofuel sectors, including enhanced incomes for farmers. This means that impacts vary strongly by region, with Brazil actually having a projected poverty reduction because of its status as a biofuel exporter.

De Hoyos and Medvedev assume a more conservative baseline scenario (biofuel adoption remaining flat at 2004 levels), therefore effectively modeling a larger shock. They find an additional 32 million people in poverty in 2010 in their more far-reaching biofuels policy scenario compared with the baseline.

Another example of this type of approach is Wiggins and Levy (2008), which was an input to the United Kingdom's Gallagher Review (UK RFA 2008). Looking at five sample countries, Wiggins and Levy found that predicted price rises, even given gains in other areas of the economy, would be consistent with poverty increases that were small in percentage terms but in India, at the extreme, would be equivalent to more than 10 million people falling into poverty.

Projections in which poverty increases on a scale of tens of millions in the medium term suggest that ActionAid's assertion that biofuel mandates may have increased the global poverty ranks by 30 million in 2008 is at least plausible and somewhat consistent with the literature. Having said that, making any calculation for a situation such as a price spike is intrinsically uncertain, and hence substantially smaller or larger suggested values for the increase in poverty would also have been plausible. While it is no doubt helpful in generating media interest to propose a single value, doing so may give an inappropriate sense of certainty.

These assessments of poverty increases are all made based on price changes from economic models or analysis of historical data from price spikes. There is also a growing literature based on anecdotes and case studies that considers the more direct negative impacts of biofuels expansion on communities, often with particular reference to the weakness of customary land rights in many developing nations. On the other hand, there is a similarly growing body of literature looking at opportunities for biofuels to support development and reduce poverty. The existence of these two bodies of work does not have to be seen as contradictory. The first tends to look at examples of relatively large-scale foreign investment in plantations, often intended for export, focusing on areas where the full prior and informed consent of local communities may not have been sought. The second tends to focus on local production for local use, smallholder engagement, and the application of good practices with regard to sustainability. In reality, examples of both positive and negative outcomes will inevitably take place, and the balance between them is the subject of lively debate. This is not the appropriate place to attempt a full review of those studies.[41]

While the exact scale of the marginal poverty increase that can be attributed to biofuels production is difficult to quantify, and the potential for bioenergy development to provide welfare gains in some cases should not be overlooked, it seems reasonable to state that biofuel mandates are likely, at the global scale, to result in non-negligible increases in poverty rates. This conclusion has led a variety of international organizations, experts, and antipoverty campaigners to call for biofuels support to be ended to reduce pressure on food markets.[42]

3.1.2. Food price volatility

It is likely that biofuel mandates have not only aggravated price rises during spikes but have contributed to systematic increases in the **volatility** of agricultural commodity markets. The higher volatility comes about because biofuel support mechanisms like mandates introduce additional **inelastic** demand to agricultural markets (biofuel mandates do not change in size when prices rise). David Laborde (2011b, p. 3) argues that, “in the short run, these rigid [biofuels] policies, by their nature, contribute significantly to price volatility and are potentially more toxic than traditional farm support or decoupled programs.” The UN Committee on World Food Security’s High Level Panel of Experts on Food Security and Nutrition made similar connections (FAO HLPE 2011, p. 40). Volatility in markets has a negative effect on welfare in addition to any impact on long-term prices. To take an extreme case, a

---

41 One starting point for further reading would be the contributions to the two conferences International Conference on Global Land Grabbing and the International Conference ‘Bioenergy for Sustainable Development in Africa’—Lessons Learnt from COMPETE.

42 For example, the UN Food and Agriculture Organization, the International Fund for Agricultural Development, the International Monetary Fund, the Organisation for Economic Co-operation and Development, the UN Conference on Trade and Development, the World Food Program, the World Bank, the World Trade Organization, the International Food Policy Research Institute, and the UN High-Level Task Force on the Food Security Crisis (2011); the Institute for Agriculture and Trade Policy and Global Development and the Environment Institute (2012); the UN Food and Agriculture Organization High Level Panel of Experts on Food Security and Nutrition (2013); The UN Special Rapporteur on the Right to Food (2011); Laborde (2011b); the Financial Times (2008); World Bank President Robert Zoellick (2008).

large price spike such as in 2008 could cause famine even if the longer-term trend was for food prices to go down.

Based on these ideas, Laborde recommends that if biofuels support is to continue, there should be a shift toward mechanisms that allow biofuel demand to respond to world market conditions. In the best-case scenario, it is possible to imagine that a reactive biofuels policy could contribute to stabilizing agricultural commodity prices in a way that more than compensated for the negative impacts of the longer-term upward pressure on prices. It is important to note that such a policy would to some extent transfer risk from food consumers to biofuel producers, and this might warrant enhancing the support given to them compared with what is available under the current set of mandate-based policies.

3.1.3. Food consumption reduction in the modeling of indirect land use change

As noted above, the extent to which a model scenario includes reductions in the consumption of feedstock for food and other purposes can be an important determinant of the indirect land use change predicted by that model. If there is a substantial displacement of food, it can significantly reduce the indirect land use change emissions.

In the models, the parameter (or rather set of parameters) that determines these effects is the food and fodder demand elasticity. This parameter tells the model what percentage reduction in demand would be associated with a certain percentage increase in price. So a demand elasticity of 0.2 would mean that a 10 percent price rise would result in a 0.2 x 10 percent = 2 percent reduction in demand, while a demand elasticity of 0.5 would imply that a 10 percent price rise would result in a 0.5 x 10 percent = 5 percent reduction in demand.

This simplified explanation of the concept of the demand elasticity is rendered a little more complicated upon recalling that the economic models solve for a variety of effects simultaneously. That means that the demand elasticity, in conjunction with other elasticities, affects the degree to which the price rises in the first place. It could therefore be misleading, in the context of analyzing land use effects, to look at the demand elasticity without also considering the crop area elasticity, for example (see section 3.5.1). In the words of the European Union's Joint Research Centre (Edwards, Mulligan, and Marelli 2010, p. 18), "elasticities cannot be compared one by one because it is often the relative size of elasticities which is important."

It should be recognized in mind that not all reductions in consumption will have the same types of welfare impacts. In some models of U.S. corn ethanol such as GTAP, the bulk of the decline in consumption is predicted to occur in the United States. Clearly, this raises issues different from a similar consumption reduction in a developing nation. AGLINK modeling of European Union rapeseed biodiesel, on the other hand, has almost all of the reduction in consumption occurring outside Europe.

There are also nonfood crop uses included in the models, such as cotton and tobacco. One might argue that increasing the price of tobacco could have positive welfare outcomes,[43] while the price and availability of cotton is certainly a less emotive issue than that of food but could still have welfare impacts.

43 While health campaigners generally welcome increased tobacco prices, if higher tobacco prices result not in reduced smoking so much as diminished income for other purchases, even this could have negative welfare consequences in some cases.

One way to investigate the importance of food consumption reductions for the carbon emissions intensity results is to run sensitivity scenarios with the models holding food consumption constant. The results of these sensitivity cases can then be compared to the central biofuel policy case. This technique effectively allows for modeling what would happen if countries introduced compensatory food subsidies to offset price rises. The Expert Workgroup on Indirect Land Use Change run by California's Air Resources Board (ARB) in 2010 discussed it as a way to avoid giving 'credit' to biofuels for cutting food consumption. While it is a useful exploratory tool, the results generated by holding food consumption constant in this way do not reflect a credible scenario, nor do they fully reflect the importance of food consumption changes in reducing the estimated ILUC emissions of a given model.

The results will not be likely to mirror the real world because, while some level of government measures in response to rising food prices is plausible, it would be highly unusual for all governments worldwide systematically to increase food security support by exactly the necessary amount to counterbalance the increasing prevalence of biofuel mandates. Even if some countries introduce such measures, given that the medium-term price rises prompted by biofuels policies will be persistent and gradual and that, at the moment, government action even in developed countries like the United States is inadequate to stave off hunger in the population, such an outcome is implausible.

These results do not fully underscore the importance of food consumption changes in the central scenarios because when food consumption is held constant, it causes prices to increase even more than in the base case. While this will indeed drive additional land expansion, it will also propel all the other model responses like crop switching and yield intensification that mitigate the effect of demand increases. Thus, the difference in emissions by holding consumption constant would be expected to be a lower bound on the contribution of reduced food consumption to avoiding land use changes in the central cases.

Hertel et al. (2010b) apply this fixed consumption approach in their GTAP modeling and note that it doubles the impact on forestry and increases overall expected land use change emissions intensity by 41 percent,[44] that is to say, about 10 grams of carbon dioxide equivalent per megajoule (gCO_2e/MJ). Laborde (2011a) similarly runs the MIRAGE model with fixed food consumption but notes that 'intermediate consumption' (the use of food ingredients by food processing sectors) can still change. For this modeling run, Laborde finds that ILUC emissions are increased by 20 percent overall—about 8 gCO_2e/MJ.

The Joint Research Centre (Edwards, Mulligan, and Marelli 2010) presents an alternative way to analyze food consumption changes. It breaks down the results of a number of models, detailing how various parameters affect the expected land use change. Figure 3.2 shows that there is a substantial variation in the importance of food consumption reductions in each model. For three of the scenarios, the percentage of feedstock from reduced food consumption is consistent with the Roberts and Schlenker (2010) result mentioned above, while the other three show smaller effects.

44 Page 12 of the report states that the increase is 50 percent, but the more detailed data on page 25 confirm that 41 percent is more accurate.

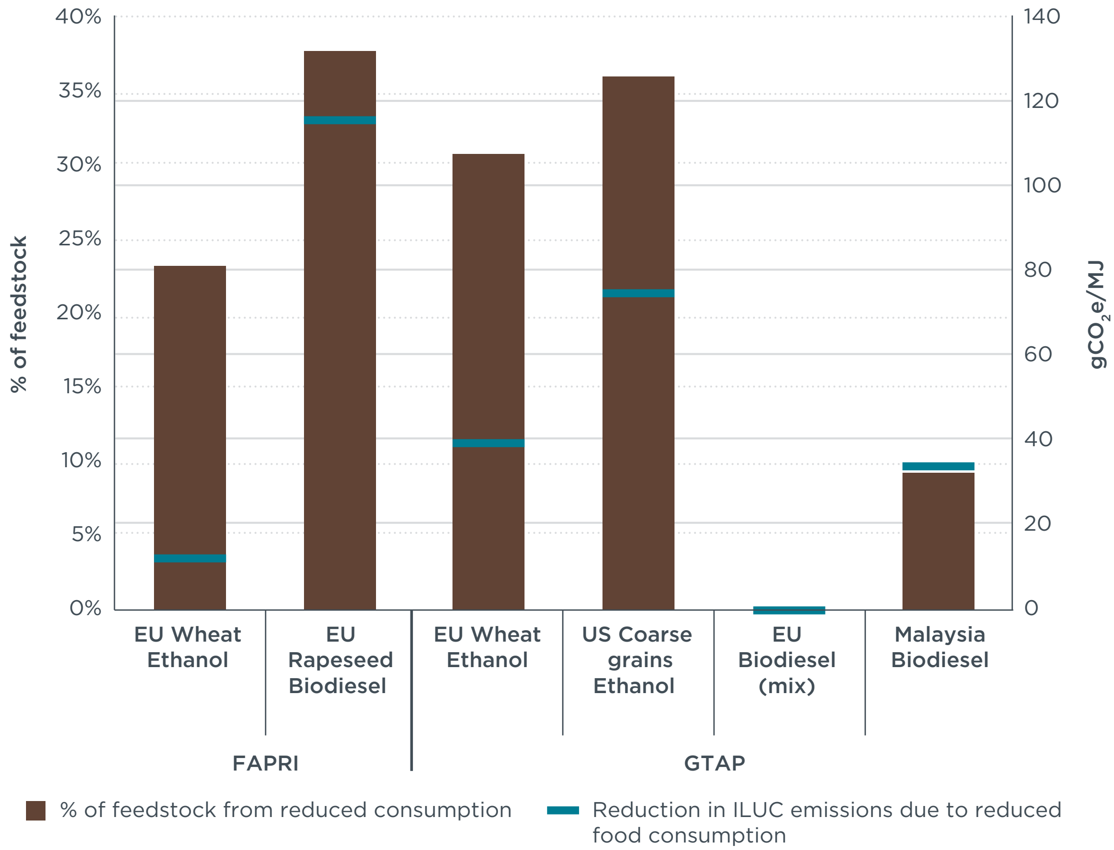

Figure 3.2 Relative importance of reduced food consumption in different GTAP and FAPRI scenarios*

* Percentage of feedstock from reduced consumption (left axis) and an approximation of the additional carbon emissions that would have resulted had this feedstock come from land expansion instead (right axis; based on Edwards, Mulligan and Marelli (2010), using a 20-year spread, or "amortization")

An interesting example is the modeling for the Joint Research Centre using GTAP for a U.S. corn ethanol scenario.[45] Using the technique of holding food consumption constant and rerunning the model, Hertel and colleagues report a change of 10 gCO_2e/MJ. In contrast, the JRC analysis attributes an ILUC reduction of 75 gCO_2e/MJ to food consumption. If the JRC breakdown is correct,[46] food consumption reduction is indeed much more important in reducing ILUC emissions than might be concluded from running the model again with food consumption held constant. Part of the difference is the amortization (how land use changes are distributed across time; Hertel et al. spread ILUC emissions over 30 years, while JRC uses 20 years), yet, even correcting for this, JRC is still assigning five times as much importance in terms of carbon emissions avoided.

45 The results of breaking down this model run should be broadly applicable to the GTAP modeling for the California Air Resources Board, as it uses largely the same version of GTAP.

46 And if it is an acceptable simplification to assume, as has been the case here, that the emissions of the land use change avoided through food reductions will be proportional to the overall ILUC emissions.

3.2. ELASTICITY OF YIELD TO PRICE

Crop yield is the quantity of a crop produced in a year on a given area. If yield is high, less land is needed to produce the same amount of a given agricultural commodity, so the assumptions one makes about yields are vital to determining the likely extent of indirect land use change.

Three important questions must be answered in order to quantify indirect land use change:

1. What is the baseline rate of yield change (i.e., what will the yields of different crops be at some point in the future if there is no additional expansion of biofuels)?
2. Will the expansion of biofuel markets affect the yield of crops on existing cultivated areas?
3. If cultivated land area expands, how will the yield on the new areas compare with yield on the existing areas?

3.2.1. Baseline yields

Crop yields and the rate at which they change have been a central question in economics for centuries. At the close of the eighteenth century, Thomas Malthus published his seminal theories on the growth of food supply and demand in *An Essay on the Principle of Population*; Malthus believed that food supply grew at a linear rate (Malthus 1798, p. 6). There are several ways to increase yield for any given crop and location. Typically, available measures include new varieties through selective breeding or genetic modification, improved farming practices such as crop rotations, increased or improved use of fertilizer and irrigation, and better pest control. A linear trend in crop production represents a sum of all these efforts. This idea of linear growth in crop production is confirmed by the historical data. In the words of the UK environmental consultancy ADAS (2008, p. 9),[47] “Despite substantial variation in yield trends between countries, the world trends for most of the major crops show a remarkably consistent linear trend.” The same point is made by Alexandratos (1999); Balmford, Green, and Scharlemann (2005); Calderini and Slafer (1998); Evans (1997); Finger (2010); and Hafner (2003). Steven T. Berry and Wolfram Schlenker argue that “the estimated yield trends are remarkably close to linear, which is consistent with steady technological progress that changes very little with changes in medium-run market conditions” (Berry and Schlenker 2011, p. 3). Figure 3.3 shows yield and price over the past 60 years for U.S. corn. While prices have varied a great deal, it is immediately noticeable that those large variations are not matched by any corresponding systematic variation in yield trends.

47 A more extensive discussion is available in Hafner (2003).

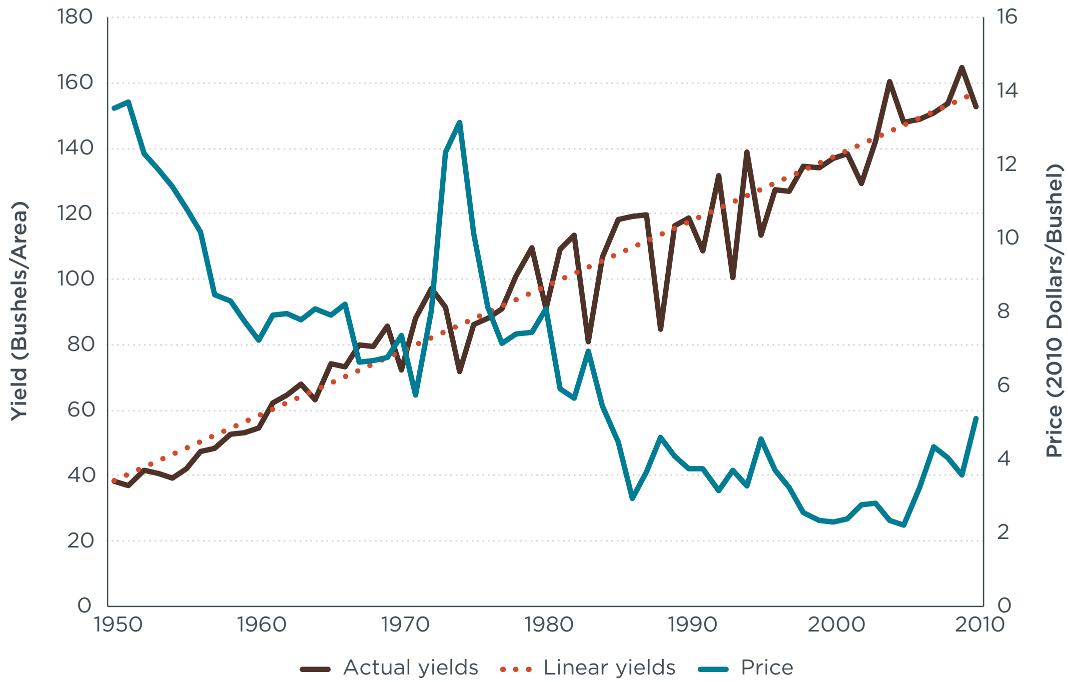

Figure 3.3. Variation of yield and price of U.S. corn over time, with linear yield trend

Based on Berry and Schlenker (2011, p. 4)

Baseline yield growth is important to ILUC modeling because if the average yield of all crops increases by 10 percent over 10 years, the land area required to grow them is reduced correspondingly. The same is true for the expansion of cultivated land required to supply a biofuel mandate.

Current yield varies not only by crop but also by the region where the crop is being grown (Table 3.1). Some of this variation reflects differences in climatic conditions; for instance, one would expect better grape yields in sunny Spain than in cloudy England. Much of it, however, reflects the different agricultural models and levels of sophistication between the developed and developing worlds.

Table 3.1. Variations in 2010 wheat yields for selected countries[48]

Country	Yield in metric tons per hectare
United Kingdom	7.7
Argentina	3.4
United States	3.1
India	2.8
Ukraine	2.7
Turkey	2.4
Ethiopia	1.8
Australia	1.6

48 Data from the Food and Agriculture Organization statistics division, FAOSTAT, available at www.faostat.fao.org.

Intuitively, one might think that developing countries would be able to achieve better rates of yield growth than countries in the developed world because the latter are likely to be closer to maximum production capacity given use of current technologies (for example, farmers in developed countries are likely already using fertilizer at high levels) and because decreasing returns to inputs like labor and capital are to be expected.[49] Nevertheless, there is no strong evidence for this sort of differential in crop yield growth. This is not entirely surprising—many of the reasons that developing world yields tend to lag developed world yields are structural and not easily overcome. Knowledge dissemination can be much more challenging in the developing world, and developing world farmers are likely to have much poorer access to capital. Some studies have also reported that farmers in developing countries may prefer to use low-yielding crops because they cannot afford fertilizer and because crops that allocate less energy to grain tend to produce more straw for animal feed (Parikh and Krömer 1985; Yevich and Logan 2003).

Looking forward, though, there may be reasons to expect that the rate of yield growth could actually dip below the historical average. There are cases in which yields seem to have hit some sort of plateau, but these are not the norm. ADAS (2008, p. 9) remarks, "Despite yield plateaus in some crops in some regions, plateaus are not generally strikingly evident in world yield trends, though reductions in the improvement rate are apparent in some crops. Overall yields have tended to increase in a linear, arithmetic, Malthusian fashion." While older studies generally supported this hypothesis of yields increasing at the same rate as in the past (Dyson 1999; Hafner 2003; Wang and Davis 1998), more recent papers have detected a slowdown in yield gains since the mid-1990s. According to these studies, yields are still increasing but at a lesser rate than before (Brisson et al. 2010; Calderini and Slafer 1998; Finger 2010; Kucharik and Ramankutty 2005; Lin and Huybers 2012). Some authors predict relatively slow yield growth in the coming decades (Cassman 1999; Evans 1997; OECD-FAO 2012). The main reason for the deceleration is that the easiest yield gains have often already been achieved. There are limited further gains to be achieved from conventional breeding, and fertilizer is already at maximum application rate in most Western agriculture (ADAS 2008; Bell et al. 1995). It is also possible (but unproven at this stage) that climate change is already starting to have a negative impact on yields (Brisson et al. 2010; Lobell, Schlenker, and Costa-Roberts 2011). Climate effects will likely strengthen over time.

Overall, it seems reasonable to assume that developing world yields will not suddenly leap to catch up with developed world levels, but nor will baseline yield increase stop in the developed world, at least not over the next decade. The evidence does, however, suggest that assuming a continuation of the underlying trend yield growth of the past 50 years may be optimistic. The lower the trend rate of yield increase, the more impact biofuel demand is likely to have on ILUC and food markets. This expectation is confirmed by sensitivity analysis for the U.S. RFS2 (U.S. EPA 2010a, §2.6.2.2). To test the importance of baseline yields to the model results, the U.S. Environmental Protection Agency (EPA) ran scenarios in which baseline 2022 yields would be 25 percent higher for corn and soy than in the central case. It found that the emissions intensity for both soy biodiesel and corn ethanol would be reduced. That said, the change was different

49 Decreasing returns means that each additional unit of labor and capital is less productive than the last. So the first laborer a farm owner hires might increase crop production by ten units, the second laborer, by eight, the third, by six, and so on. The same applies to capital (e.g., a $100,000 combine harvester may get its purchaser 80 percent of the benefit that a more expensive $200,000 model would).

for the two fuels—other effects in the model damped the benefit for corn but amplified it for soy. This is a useful reminder that because of the complex interlinkages in the agricultural models, just as in the real world, effects do not always act in a simple linear way as one might initially expect.

3.2.1.b. Climatic effects on yield

One important caveat regarding projected baseline yield trends is that they assume the effects of climate change on crop yields will be relatively limited, at least in the period to 2020. It is worth bearing in mind, though, that some studies suggest that climate change may already have significant impacts on crop yields.

Parry et al. (2004) carry out modeling of global yields based on various climate change scenarios from the UK Met Office's Hadley Centre for Climate Prediction and Research. They find that yields are likely to be reduced by climate change but that these reductions will be very small in the near term—the effects in 2020 are characterized as "within historical variations." Climatic stresses are predicted by Porter and Semanov (2011) to reduce wheat yields and increase yield variability, and several other studies (e.g., Rahmstorf and Coumou 2011; Wergen and Krug 2010) suggest that extremes in weather have already increased because of global warming. Berry and Schlenker (2011, p. 4, Figure 1) provide an illustration of the importance of climate in determining yields. They find that the variation in weather can be used to explain almost all variation in annual yields when combined with an underlying linear yield trend. At some point in the coming decades, if not already as some studies suggest, climate change probably will start making a significant impact on yield growth. There is not at the moment compelling evidence that the general trend will be substantially reduced by 2020, but concern about climate change is another reason that it might be appropriate to consider the historical rate of yield change to be an upper bound on expected baseline change for the coming decades rather than a most likely scenario. As the Joint Research Centre (Marelli, Mulligan, and Edwards 2011, p. 40) puts it, "If the extreme weather experienced globally in 2010 is an indication of what's coming in the next decade and beyond, existing models will tend to under-estimate yield in some areas and over-estimate yields in far more locations."

3.2.2. Price-induced yield increase

There are two ways that one can conceive of price-induced yield increase. It can mean the change[50] in the yield achievable on a given piece of land for a given crop when demand, such as that for biofuel, increases. This might also be referred to as intensive yield increase (yield increase due to intensification of production on existing areas). It is also possible to draw a broader definition of the term 'price-induced yield change' to include all yield effects, capturing any change in average yield resulting from bringing new land into production. The broadest possible definition could even include yield change from changing which crops are grown (see section 3.3.1 on crop switching).[51] The overarching factor 'elasticity of yield to price' includes both intensive yield increase and effects due to lower yields at the margin of production, but excludes crop switching. For this subsection, however, the focus will be only intensive yield change.

50 In general an increase, but complex market dynamics could lead some prices to fall because of increased biofuel use (e.g., increased supply of wheat distillers grains might reduce the price of some other livestock feed ingredients), in which case presumably yields could drop for those ingredients.

51 For example, oil palm is a high-yielding oilseed, so if palm oil expands relative to rapeseed oil, the average oilseed yield will rise, while if rapeseed oil expands faster than palm oil, the average oilseed yield will fall.

Within intensive yield increase, one can draw a further distinction between two types of effect. The Joint Research Centre (Edwards, Mulligan, and Marelli 2010) refers to these as reversible and irreversible yield increases.

3.2.2.a. Reversible yield increases

There are various actions that can be taken in the short term to increase yields but that which will have no long-term implications. The most important of these is to increase application of agricultural inputs, in particular, fertilizers. Other reversible inputs might include labor (refer to economic models like MIRAGE in which 'factor intensification' allows labor and capital to be substituted for land). As long as higher prices prevail, higher fertilizer application rates, etc., will presumably be maintained,[52] and hence the yield increases will persist as long as the prices do. However, if prices fell, yields would return to the base level.

While these reversible yield increases offer the potential to reduce ILUC emissions, there may be negative consequences to increased fertilizer use. The USDA notes (Marshall et al., 2011, p. 21) that "increased nitrogen application may result in increased direct N_2O emissions, and more intensive farming practices may result in increased erosion and decreased soil carbon sequestration." In general, the emissions implications of increased fertilizer use have not been included in modeling of indirect land use change, even when increased fertilizer use is part of the model response to rising biofuel demand. The Joint Research Centre (Edwards, Mulligan and Marelli 2010, p. 106, Figure 26) points out that, especially where rates of fertilizer use are already high, the increased emissions of nitrous oxide (N_2O) stemming from increasing fertilizer application might actually cancel out any benefits from avoiding land use change. Fertilizer emissions can be accounted for in both the Food and Agricultural Policy Research Institute–Forest and Agricultural Sector Optimization Model (FAPRI-FASOM) ILUC modeling for the EPA and the Green-AgSim model used for some other FAPRI modeling.

3.2.2.b. Irreversible yield increase

As well as temporary yield-enhancing measures, there is the possibility that elevated agricultural prices spur longer-term investments by farmers (such as upgrading machinery) and agricultural research investment. The High Level Panel of Experts (HLPE) of the UN's Committee on World Food Security (FAO HLPE 2011, p. 27) paints a picture of cycles in which periods of high agricultural investment improve productivity, leading to periods of reduced prices, resulting in a reduced level of investment, eventually leading to a food crisis that puts pressure on prices, spurring new investment and restarting the cycle.

If one accepts this analysis of the global food market, then price spikes such as the one in 2008 (driven, as discussed above, at least somewhat by biofuels) can lead to a resurgence in investment and (in due course) to increased commodity production. Even without espousing a cyclic view of agricultural investment, it is still possible to make the case that increased prices will provide incentives for companies to invest more heavily in research, as well as an incentive to producers to adopt these innovations. Once innovation has occurred, it would not in general be rational to roll it back,[53] so these yield gains are seen as irreversible.

52 It is a simplification to talk in terms of feedstock prices only, as input prices themselves may vary.

53 There will always be exceptions, in particular, if a new technology has short-term benefits but unforeseen long-term drawbacks.

Baseline yield assumptions can make a difference to expectations for ILUC, but the more important question is whether yields are forecast to rise across the board, given increased demand for biofuels, more than would otherwise be the case. The idea is that, with increased biofuel demand, commodity prices will rise, and farmers and others in the agricultural industry will be able to invest more in raising yields. If the response of yields to prices is large compared with reductions in demand or increases in area under cultivation, then it is logical to conclude that biofuel demand can be met without driving large ILUC emissions or precipitating substantial hunger.

As well as motivating farmers to make shorter-term productivity gains, many commentators[54] also argue that biofuels can be a key driver of investment in agricultural research and development, helping to advance longer-term, persistent yield gains. In the strongest version of this hypothesis, rather than inducing large-scale land use change and creating competition with food, biofuels can push forward innovation and productivity gains that in the longer term help to bring food prices back down. This reflects to some extent the cyclic vision of agricultural investment from the HLPE (FAO HLPE 2011).[55]

3.2.2.c. The argument against a strong connection between yield and price

The narrative case for significant price-induced yield increase seems reasonable,[56] and there is a degree of agreement in the field about price driving yields to some extent, but it is possible to make counterarguments. Bouët et al. (2009, p. 17) observe, "One recent analysis [Liu and Shumway 2007] concluded that relative price changes have not encouraged innovation in U.S. agriculture in the last 40 years. The paper concludes, 'This finding cautions against the efficacy of policies based on the premise that price signals alone induce efficient technical change.'" Liu and Shumway (2007) merely show no evidence for the induced innovation hypothesis, but it is even possible to argue that high prices actually reduce the rate of innovation uptake. Ecofys (2009), as quoted by the European Commission's Directorate-General for Energy (EC DG Energy 2010, p. 96) writes, "The adoption of new technologies by farmers has been incentivized by declining producer prices. Falling crop prices forced farmers to reduce input costs by adopting new technologies in order to maintain a sufficient margin. The resulting growing output or reduced input costs again reduced crop prices, forcing farmers to reduce input costs further."

From this point of view, instead of increasing prices promoting higher investment in research and development, one might actually expect diminished pressure on farm margins to reduce the incentive to innovate or adopt innovation. In conclusion, Ecofys argues that, "while some support for the theory of price-induced innovation can generally be found, other factors than price also play an important role," and that "innovations may also be input saving without increasing yields per ha." The hypothesis that biofuel demand will induce some sort of green revolution is certainly anything but clearly confirmed from the existing literature.

It is also not universally accepted that farmers will successfully intensify crop yields in response to price. Berry (2011, p. 8) comments, "There is a long tradition in agricultural

54 For example, the European Commission's Directorate-General for Energy (EC DG Energy 2010).

55 Note that the HLPE itself does not itself represent biofuels as a beneficial intervention in agriculture spurring new innovation but in fact argues that "limiting the use of food to produce biofuel is the first objective to be pursued to curb demand" (FAO HLPE 2011, p. 40) and calls for mechanisms to relax biofuel demand during periods of tight supply.

56 The European Commission's Directorate-General for Energy (EC DG Energy 2010) wrote that "it makes little sense from the point of view of economic theory to argue that yields are independent of demand."

economics (dating back to Wright [1928] and Nerlove [1956]) that takes as obvious the notion that almost all of the price-elasticity of supply comes from land-use rather than yield. In these cases, changes in yields are treated as determined by technological change (in the long run) and by weather (in the short run)." Roberts and Schlenker (2010, p. 8) contend that "if yields were responsive to price levels, we would observe that yield shocks are correlated between various countries in a given year, as all countries face the same world price." Such correlations are not seen in the historical data. Berry (2011, p. 14) also notes that farm-level behavioral studies (e.g., Hertel, Steigert, and Vroomen 1996) have been unable to find evidence of a positive yield response to price.

While the microeconomic arguments for anticipating that yields will diminish as prices rise are interesting, and likely have validity in some instances, this report accepts the majority opinion that a positive elasticity of yield to price seems likely. Still, the evidence seems inadequate to say with confidence that this elasticity will be large compared with the elasticity of area expansion. In the words of Berry (2011, p. 18), "There is much resistance to a literal value of zero for the yield-price elasticity. There is evidence that farm inputs (such as fertilizer use) respond to prices, which is consistent with some positive value for the yield elasticity." This view has some merit, although it does not support any particular positive value.

3.2.2.d. Yield and public investment in agriculture

There is more consensus in the literature for a separate conclusion—that publically funded research and development has historically been a key driver of innovation. Ecofys (2009), as quoted by the Directorate-General of Energy (EC DG Energy 2010, p. 96), maintains, Publicly funded R&D in agricultural technologies has been an important source of the technological innovations that made the dramatic yield increases of the past decennia possible. Publicly funded R&D played a major role in the early advancements in the developed world in the first half of the 20th century and also the transfer and adoption of these technologies to developing countries in the Green Revolution was largely made possible by not-for-profit institutions.

DG Energy puts forward the argument that a decline in annual yield improvements over the past 30 years corresponds to reductions in public agricultural investment. In the context of the ILUC and biofuels debate, it is worth considering that many commentators in favor of government biofuels support have argued for a strong effect of biofuel mandates on technological advancement, presumably because of increased commodity prices. While higher agricultural prices may have a part to play in boosting R&D spending over the coming decades, direct government investment would be a much more certain way to achieve these goals, as well as potentially cheaper. For example, the cost to the U.S. government of the volumetric ethanol excise tax credit (VEETC), an ethanol blending credit, in 2010 was about $6 billion. This is six times the annual R&D budget of the biotechnology firm Monsanto.[57]

3.2.2.e. Effect of price-induced yield change on ILUC

Assumptions about price-induced yield increases in biofuel feedstocks can have a substantial impact on the magnitude of indirect land use change and on informed opinion about the benefits or drawbacks of biofuel mandates. Because biofuels still represent

57 The Monsanto website, as accessed in April 2012, states, "Monsanto invested more than $980 million last fiscal year researching new tools for farmers." http://www.monsanto.com/investors/pages/corporate-profile.aspx

only a relatively small fraction of the total market for most feedstocks, relatively small increases in average yields across all production of that feedstock could make a large difference in terms of net land demand.

To consider a simplified example, imagine having 100 hectares of feedstock production for food, with a yield of one metric ton per hectare. Subsequently introducing demand for 10 tons of feedstock to produce biofuels would require 10 extra hectares if nothing else changed. However, if price increases stimulated an increase in yields of 10 percent over 10 years, then the new yield would be 1.1 tons per hectare, meaning that total production would rise to 110 tons per annum, and 10 tons would be available for biofuels with no extra land requirement.

Whatever the crop, the most important question in ILUC estimation is always how the yield response compares to other responses—whether it will be greater than, comparable to, or less than the land area response (discussed in further detail below) and consumption response (discussed above). The economic parameter used to quantify the strength of the yield response is the 'yield on price elasticity.' The example in the previous paragraph required that the cultivated area and the food demand be totally inelastic to price. If the area elasticity were comparable to the yield elasticity, then one might expect to see, instead of a 10 percent yield increase, a combination of a 5 percent area increase and 5 percent yield increase to meet the additional demand,[58] with price only needing to increase half as much. If food demand elasticity were also comparable, to accommodate the same demand increase as in the example above would necessitate closer to a 3 percent increase in yield, 3 percent reduction in other demand, and 3 percent increase in area cultivated, with only one-third of the price rise.

3.2.3. Yield on price elasticity

In the modeling of indirect land use change, the important parameter for intensive yield change is the yield on price elasticity. This tells how much average yield on a given piece of land is likely to increase in response to a given increase in agricultural prices. For instance, a yield on price elasticity of 0.1 would imply that for every 10 percent rise in prices, yields probably would rise by 1 percent. A yield on price elasticity of 0.2 would imply a 2 percent increase in yield for every 10 percent increase in prices. If the yield on price elasticity were much greater than the area on price elasticity (see section 3.5.1), yield increases would in most cases be sufficient to avoid major indirect land use changes. If they were about the same, ILUC would be reduced but still significant. And if the yield on price elasticity were lower or much lower, ILUC emissions would be affected relatively little.

The simplified example offered in the previous subsection can serve to explore what yield on price elasticities mean. Starting with 100 hectare, 1 metric ton per hectare, adding 10 tons of demand, it was demonstrated that a yield increase of 10 percent would avoid the need for land use change. With a relatively high yield on price elasticity of 0.2 (what values seem plausible is discussed in more detail below), achieving such a shift in yield would imply that prices had to increase by 50 percent (0.2 * 50 percent = a 10 percent yield increase). This level of price increase is quite high and (remembering the food-versus-fuel discussion above) would certainly have welfare impacts.

58 If the extra area cultivated also experienced the small yield increase, in fact, slightly less than 5 percent increases would be necessary for each.

To take a real-life example, currently, about 40 percent of the U.S. corn harvest is used for ethanol production. Using a first-order assumption that distillers grains are returned to the feed market, reducing the net corn demand, this is like having 27 percent of corn committed to biofuel. If the yield on price elasticity of corn in the United States were 0.2, then to get all of the biofuel from yield increase would necessitate a price increase for corn of 135 percent and negligible elasticities of food demand and area to price. If the yield on price elasticity of maize were only 0.1, a price increase of 270 percent would be needed.

There is also a question surrounding the time frame for assessing elasticities. It is generally agreed among economists that short-run elasticities are likely to be lower than long-run elasticities—that is, the response to price increases in the short term will be less than the long-term response if those increases are sustained for several years (Edwards, Mulligan and Marelli 2010, p. 110). For the modeling of ILUC, responses in the medium term (more than a year but less than a decade) are most relevant, but econometric analysis in the literature is generally short term. Berry (2011, p. 7) argues, "The long and short-run distinction is particularly important for land-use elasticities. It is costly to transform land from one use to another and so land is likely to be put into a new use only if an economic change is likely to persist." Other authors have made the same case but have focused on yield elasticities; the Air Resources Board of California (ARB 2011, p. 4) makes the case that "the long-run responsiveness of yield to price will be greater than the short-run response if there are lags in the adoption or development of new management practices or seed varieties." It is unclear based on the existing literature and consideration of the arguments whether the difference between short- and long-run elasticities should be greater for area or for yield. In each case, it is easy to imagine actions that would only make sense if one expected a long-term price increase, but it is also easy to identify actions (bringing fallow land into production, changing rotations, applying extra fertilizer) that would be rational in the short term. One should therefore be wary of any argument that short-run values ought to be inflated for one elasticity but not the other, and similarly skeptical of any modeling that uses short-run analysis for one type but long-run analysis for the other without correcting.

Yield on price elasticity is important not so much in itself but for how it compares with other elasticities, in particular, the elasticities to price of overall cultivated area and of food demand. For yield to be the dominant response to an increase in feedstock demand, yield on price elasticity needs to be much larger than area on price elasticity and consumption on price elasticity, so that, even for a strong change in commodity prices (e.g., a 135 percent increase in corn prices), consumption and the cultivated area do not change much. Otherwise, expansion of the area under cultivation and consumption would act as safety valves for prices, preventing them from rising to the levels necessary to induce large yield increases.

Regional variations in yield response?

Like the rate of baseline yield change, the size of the yield response to price is liable to vary between regions, depending on the characteristics of existing agriculture and issues like access to finance (exogenous considerations like interest rates may affect the willingness and ability of farmers to invest). Farmers in developing countries may have more capacity to respond to price incentives by increasing their fertilizer use because they are less likely to already have reached maximum levels of fertilizer application. In the words of the Joint Research Centre (Edwards, Mulligan, and Marelli 2010, p. 104), "one can expect that yields respond to prices more in developing countries where yields are further from the technological limits (and there are good returns to applying more fertilizer, for example)." Militating against this potential flexibility are issues like access to capital and access to information that render farmers in the developing world less able to take advantage of increased prices. More generally, there are all sorts of reasons for farmers in different places to be more or less responsive to price.

What are appropriate levels for yield on price elasticity? A review paper by Keeney and Hertel (2008, p. 20) of econometric evidence on price-induced yield elasticity for U.S. corn suggested a net yield on price elasticity of 0.25.[59] A working paper by Huang and Khanna (2010, p. 15) looked again at U.S. data and suggested a value of 0.15 for corn, 0.06 for soybeans, and 0.43 for wheat.

The value of 0.25 has been adopted in the GTAP economic model but has been vigorously disputed by Berry (2011). Berry argues that Keeney and Hertel mischaracterize the results of the studies they consider and that the results reviewed are based on inadequate econometric analysis in which the authors do not demonstrate a causal relationship between price and yield.

A more robust analysis of the econometric data is provided by Berry and Schlenker (2011), using the technique of instrumental variables.[60] This technique allows economists to distinguish the market effects of shifting supply to be separated from the market effects of shifting demand; for ILUC, responses to increases in demand are what matter. Using this technique, one should be able to confidently distinguish causal relationships from other correlations. For the significant majority of regions and crops considered,

59 "Net" yield on price elasticity is specified because, in the econometric evidence, it is difficult if not impossible to distinguish one type of yield effect from another. When one investigates the historical evidence of average corn yields, the data include both what is described here as the price-induced yield response and also any yield effects attributable to expanding agriculture onto new (and potentially less fertile) areas. This second-order type of effect is discussed in the next section—for now, it is simply important to understand that the net elasticities calculated from the historical data may be considered to be lower bounds for the "pure" yield effect.

60 An instrumental variable is a variable that drives either supply or demand but that is believed to be completely independent of both. The classic example is weather, used as an instrumental variable for supply. Weather is assumed to be independent of agricultural supply and demand but strongly affects supply (because if weather is bad, production falls). One could therefore use weather as an instrumental variable determining a supply shift and use it to analyze the elasticity of demand (how do consumption levels and prices vary the year after an extreme weather event, for instance).

Finding an instrumental variable for demand is more challenging. Roberts and Schlenker (2010) suggest the innovation of using lagged weather as an instrument for demand. The idea here is that stock levels will be low following a year of bad weather and high following a year of good weather. Conversely, demand will be higher following a bad year as people try to replenish stocks. The weather in, say, 2008 can serve as an instrumental variable for demand in 2009, when stocks will need to be replenished if 2008's weather was bad.

Berry and Schlenker find that the net historical yield on price elasticity is low, and not significantly statistically different from zero. For Brazilian soy, they find evidence of a nonzero positive value, but in China for several crops there is actually a negative elasticity (yields and area cultivated apparently both fall in response to high prices). In particular, they find strong econometric evidence that the yield on price elasticity in general is substantially lower than the value of 0.25 proposed by Keeney and Hertel—their estimate is that it is 0.06 or less.

In contrast, as discussed below in section 3.4, Berry and Schlenker do find significant elasticity of area cultivated to price for the world as a whole, with larger area elasticities for several important regions. Huang and Khanna (2010) similarly find higher area elasticities.[61]

When considering these yield results, especially from the point of view of modeling indirect land use change, it is important to recognize that the econometric analysis can only determine 'net yield' responses, so the results could imply either that both price-induced yield effects and marginal yield effects (see below) are small or that both are significant but that they cancel each other out (this hypothesis is advanced as a simplification by Searchinger et al. 2008, p. 2). An additional problem when looking for evidence of yield responsiveness to price in historical data is that for much of the past century agricultural returns have been stabilized by government intervention. In this context, it can be argued that it is economically rational for farmers to invest at a steady rate, consistent with the observed linear rate of yield growth. More recently, agricultural decision-making has grown increasingly market-led as agricultural subsidies become a less important part of farm incomes. In this situation, it might be reasonable to expect a stronger response of yields to fluctuating prices. But with a relatively short period of data to examine, it is difficult to come to any conclusion.

The central question here is whether price-induced yield change tempers the ILUC effects of biofuels policies to a degree that allows them to be effective as climate mitigation strategies. The following three points should inform the answer to this question:

1. The historical econometric evidence for a strong *long-run* (over several years) response of yield to price is not compelling. Figure 3.3 neatly illustrates the apparent lack of a correlation between volatile prices and a steady rate of underlying yield change. In order for yield to be the dominant response to price in the next decade compared with expansion of area under cultivation, it must be much more responsive to price in the future than it has been previously.
2. The historical econometric evidence on the *short-run* elasticity of yield to price (the year-to-year response) shows evidence, however weak, for a small positive value. Most of the literature on the subject is explicit in averring that yields are fairly unresponsive to prices (Berry 2011). It is possible to make microeconomic arguments that economically rational farmers should raise yields when prices are high and that this response should be stronger now than in the past, but this effect is not apparent in the data. Using a relatively high value is therefore a matter of microeconomic expectation or optimism rather than evidence.
3. Most studies that consider both yield and area elasticities find that area is more responsive than yield. Indeed, Roberts and Schlenker (2010) argue that area expansion is the only significant supply-side response to price. Any ILUC model

61 They find an overall area expansion elasticity to price of 0.257, higher than the yield elasticities for soy (0.06) and corn (0.15). Wheat, on the other hand, came out as more yield responsive than area responsive (0.43), but it is a less important crop in the U.S. context.

> that expects more feedstock to be supplied through yield increase than area expansion must be based on arguments that in the future yield responses will be greater than in the past and that area response will be less.

Overall, while yield change could provide an important moderating influence on ILUC, the evidence is not strong enough to expect that it will be a larger one than area expansion—indeed, there is a lack of convincing evidence that it will make an important contribution at all.

3.2.4. Cropping intensification

As well as the possibility that price signals would lead farmers to increase crop yields, it has been argued that price could lead farmers to increase the *intensity* of cropping, that is, to increase the number of crops harvested in a single year. This argument was put forward in the report of the Elasticity Values Subgroup of the Air Resources Board's Expert Workgroup on Indirect Land Use Change (ARB 2011, p. 5). That report, surveying the literature on yield on price elasticity, observed, "The overall conclusion of these studies is that the short-run response of yields to crop prices is quite inelastic." The group notes, however, based on results presented by Babcock and Carriquiry (2010) in work performed for the U.S. National Biodiesel Board, that this ignores the possibility that double cropping will increase in response to high prices. Babcock and Carriquiry note in particular that the area of double-cropped soybeans in the United States increased at the same time as the food price spikes of 2007/2008. They also point out that in Brazil, an increase in the planting of corn after soy has been harvested, although in this case there seems to be a more general adoption of the practice and less of a correspondence to increasing price. For the United States, Babcock and Carriquiry argue that the potential to double-crop could be implicitly incorporated in models by raising the yield on price elasticity. For GTAP they suggest that overall soy yield elasticity should be increased by about 0.08 to account for the option to double-crop, while for Brazil they recommend an increase in elasticity of 0.24.

As with other yield-related responses to price, an increase in double cropping seems plausible—but, as Berry (2011) remarks, "the authors do not present formal statistical or econometric evidence about double-cropping," and "the Babcock and Carriquiry anecdote is about one price increase in one country for one crop."

The European Commission's Directorate-General for Energy (EC DG Energy 2010) also discusses cropping intensity. It notes that in Food and Agriculture Organization (FAO) data an intensification of double cropping would likely be accounted for as an increase in harvested area; that is, it would look like a land use change in the data and might lead to an overestimation of land area elasticity in analysis of that data.

The apparent link between the price spike from 2006 to 2008 and an increase in double cropping is interesting, but the data are so narrow that it would likely be possible to find an alternative region and crop combination for which the picture would look strikingly different. Without a more thorough analysis of the relationship between double-cropping decisions and price/returns, it is premature to allow assumptions about double cropping to form a major plank of the basis for biofuels policy. For the modeling, it would seem appropriate to treat this effect as small compared with expansion of cultivated area, although including it as a sensitivity test could be instructive. Certainly, if policies could be constructed that actively favored double cropping, they might help to reduce the likely impact of ILUC emissions.

3.2.5. Yield at the margin of production

As just discussed, when demand for biofuels increases, it is probable that yields on existing agricultural land rise, or at least they are unlikely to fall. But as well as raising their production by intensifying cultivation on current lands, farmers can expand the land area they cultivate. Will newly converted lands have a higher, a lower, or the same yield as the average for land already under cultivation in a given region? Marshall et al. (2011, p. 21) contend, “In most regions, existing crops are already on the most productive agriculture land, so yields on newly converted lands would likely be lower than on existing cropland.” This is because production costs increase with decreasing yield (Marelli, Mulligan, and Edwards 2011, p. 37). Especially in regions with well-established agricultural economies, any remaining natural land has probably not been cultivated for a reason: it may be too dry, have poor soils, or be on a slope. If there is potentially highly fertile land that has not been cultivated yet, there may be other reasons, such as lack of infrastructure or legal barriers; higher prices might not overcome those other barriers. The Directorate-General for Energy (EC DG Energy 2010, p. 17) points out that “if converted land has a lower yield than ‘existing’ (already cultivated) land, more land will need to be converted.”

One approach to estimating yields on new land is to compare yields on existing cropland with those on land that has been set aside because of declining production in Europe and the United States. It is assumed that the land that is set aside in response to these programs is representative of the land that will be brought back into production as prices rise. Love and Foster (1990, p. 273) note that “farmers take their least productive land out of cultivation in order to meet any land diversion requirements for program benefits. This is a widely accepted belief regarding farmer behavior supported both by theoretical and conceptual work.” Despite this, some studies have failed to find strong evidence for large yield disparities, at least at the farm or county level.[62] This contrasts with studies that find a substantial effect at the national or regional level. According to Keeney (2010, p. 5), farmers may face various “limitations on unit-by-unit decision making” restricting their ability to idle or re-utilize land selectively based solely on yield potential. These include the likelihood that the most (or least) productive pieces of land may not be contiguous, increasing the cost overhead from selective idling. This does not rule out a significant marginal yield effect at the aggregate level, however. Hoag, Babcock, and Foster (1993) suggest[63] that, at the aggregate level, there may be a tendency for less productive farmers to set aside land before any others do. That would explain the observed increase in average productivity when land is set aside, even in the absence of a correspondingly large effect on individual farms.

For set-aside programs as a whole, Keeney finds a range in estimates of the ratio of marginal to average yield from about 0.5 to 0.9.[64] This is consistent with the Joint Research Centre (Edwards, Mulligan, and Marelli 2010, p. 102), which finds a ratio of 0.65 based on statistical analysis for Europe and an implied ratio of 0.71 based on UK data from the 1990s. It is also consistent with a finding by Ogg, Webb, and Huang (1984) that idled land in U.S. set-aside programs in the 1970s had soil types 65–95 percent as productive as the national average.[65] Keeney concludes, “We would expect marginal land to a crop to be no more than ninety percent as productive as that previously planted.”

62 Hoag, Babcock, and Foster (1993); Weisgerber (1969).

63 Referencing data in Weisgerber (1969).

64 Weisgerber (1969); Ericksen and Collins (1985); Love and Foster (1990); Norton (1986); Ash and Lin (1987).

65 According to Keeney (2010).

In contrast, analysis of UK and European set-aside provisions by the UK Department for the Environment, Food and Rural Affairs (UK DEFRA 2001, pp. 18–19) found minimal evidence of a low marginal to average yield ratio. The department suggests that the discrepancy between its findings on the European experience and results from Love and Foster (1990) for the United States might be a consequence of European farmers setting aside plots in rotation, so that in Europe lower-yielding land was not so systematically removed from production as it was under the U.S. scheme.

The literature on yield 'slippage' in set-aside programs is probably more relevant to relatively small changes in land use than to major changes in agricultural priorities. If a much more significant change is in the offing, for instance, a major expansion of a commodity crop export from Africa or a major resurgence of agriculture in the former Soviet Union, those agricultural decisions would be less comparable to the ones made by farmers under set-aside policies. It is not immediately clear whether cases of large-scale land expansion would result in a lower marginal to average yield ratio (perhaps because expansion could occur in areas that are generally less promising than those already cultivated) or a higher ratio (perhaps because land use has been better optimized near existing farms, while there are still opportunities to find good unused land further afield).

Several commentators point out that in some developing countries (such as South America and Africa) agricultural systems have had less time to become optimized, large areas may be available for expansion, and cropland choices may have been determined as much by location and accessibility as by land quality. In that case, one might not expect to see systematic differences between the yield on current cropland and marginal cropland. In some circumstances, because unsustainable agricultural practices may degrade cropland after a number of years, it is possible that new land may even be more productive than existing cropland. The results of set-aside analysis may not be well suited to all countries. The Brazilian research institute ICONE ('The Institute for International Trade Negotiations') has made a case that sugarcane yields on new land in Brazil may initially be lower than average yields but that once technology is adapted to local conditions, average yields can be attained (Marelli, Mulligan, and Edwards 2011, p. 27). However, ICONE also reported that yields on new land may decline over time by as much as 30–50 percent. Thus, while it is possible that the difference in yields between new land and existing cropland may be smaller in Brazil than in Europe and the United States, there is clearly variation all the same.

The Joint Research Centre (Edwards, Mulligan, and Marelli 2010, p. 102) suggests a narrative in which not only is newly converted land less fertile than the average land already in production, but crops are bounced sequentially into less suitable areas, creating a compound effect (Figure 3.4). The theory is that if more wheat is demanded for ethanol, wheat cultivation will not itself expand into new areas but will instead displace barley from the best-yielding barley areas (which will still be less suitable for wheat than current wheat fields). Barley in turn displaces rye, with lower yields for the newer barley acreage than the average, and finally it will be the rye that expands onto entirely new land, again with a lower than average yield. If this is indeed the pattern of expansion for a given crop and region, then one would expect that the observed ratio of marginal to average yield for a single crop would represent a maximum for the effective productivity of new land.

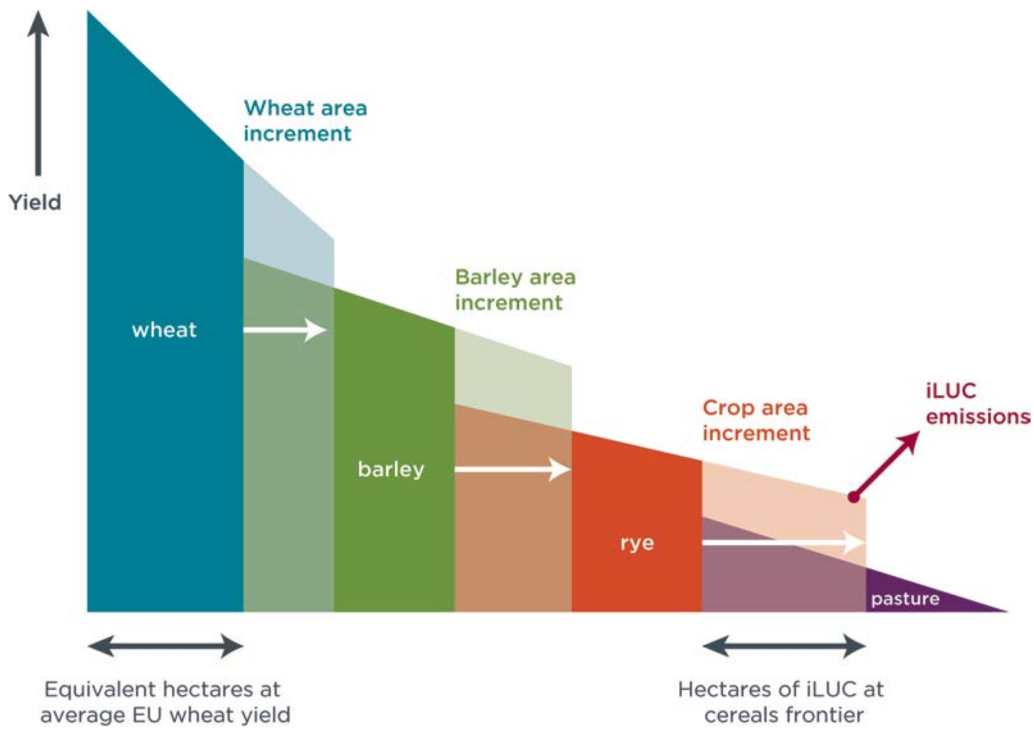

Figure 3.4. Schematic based on Edwards, Mulligan, and Marelli (2010, Figure 24) showing an ILUC multiplier effect through successive crop displacement

The possibility that, as the JRC suggests, yields will vary as crops are shifted is covered in some of the economic models. GTAP and MIRAGE both assume that the rent for a given piece of land can be used as an indicator of productivity. Hence, if a crop expands onto a low-rent land area, the yield would be lower than for a higher-rent area. Hertel et al. (2010a, p. 227) observe that, for GTAP, "in the United States, this expansion results in a decline in average coarse grains yields as maize production expands into land less suited for maize."

In the modeling of indirect land use change, the treatment of yield at the margin of production thus varies from assuming that yields are identical to using set ratios to, in recent GTAP versions, employing a relatively complex modeling assessment of global land productivity. In EPA's modeling for the RFS2, yields on new land within the United States were assumed to be exactly the same as yields on existing cropland, while internationally they were assumed to be only 2.3 percent lower than on existing cropland.[66] Al-Riffai, Dimaranan, and Laborde (2010, p. 22), using IFPRI-MIRAGE in modeling for the European Commission, assumed marginal yields to be only 50 percent of existing yields, except in Brazil, where they assumed marginal yields of 75 percent. The subsequent modeling by Laborde (2011a) modified this assumption to 75 percent for all countries.[67] Prior to 2010, GTAP assumed marginal yields to be 66 percent of yields on existing cropland[68] in all regions.

66 According to the Oregon Low Carbon Fuels Standards Final Report (Oregon Department of Environmental Quality 2011, Appendix G, p. 12), available at http://www.deq.state.or.us/aq/committees/docs/lcfs/reportFinal.pdf.

67 As documented in Laborde and Valin (2012).

68 Tyner et al. (2010, p. 58).

More recently, in new ILUC modeling work for the U.S. Department of Energy's Argonne National Laboratory, Tyner et al. (2010) detail the introduction of a more sophisticated approach to estimating the marginal to average yield ratio. This new system uses the Terrestrial Ecosystem Model (TEM) to calculate spatially explicit values for expected rates of plant growth in each region. These net primary productivity (NPP) values, which are based on suitability for growing corn,[69] can then be compared for land currently cultivated and land considered available for conversion. The ratio of the NPPs is used in the new GTAP modeling as the local marginal to average ratio. This approach produces a wide range of yield differentials depending on the region, but on the mean, marginal yields are only 10 percent lower than yields on existing cropland.[70] This result is consistent with the high end of the range identified by Keeney (2010).

The TEM analysis can be questioned, however, for several reasons. For one, it is not immediately clear that the expected NPP (total growth) of a plant is a good proxy for yield; yields could potentially be disproportionately lower on low-quality land.[71] Next, corn yield may not be a good proxy for expected yield of other major crops, all of which are C3 plants (C3 and C4 refer to different biological processes plants use to fix carbon from the atmosphere—corn is a C4 crop). The TEM does not include irrigation, and so GTAP makes a simple correction: all marginal yields are decreased by 10 percent, and no ratios or marginal to average yields above 1 are allowed. The Joint Research Centre (Marelli, Mulligan, and Edwards 2011, p. 39) also cautions that the resolution of TEM may be inadequate to fully capture many factors. It argues that, "The drivers of suitability, such as soil type or climate, in agricultural land may be inhomogeneous within a 2500km grid cell... By ignoring all other sources of yield variation (like variation on individual farms, variation due to levels of competence and investment, and variation between farms), TEM might underestimate the difference in average and marginal yield."

These assumptions (NPP as a proxy for yield, corn as a model crop, the irrigation correction, resolution) all introduce a degree of uncertainty into the model. The analysis has also been criticized (Berry 2011, p. 22) because it necessarily ignores economic considerations in land selection. Berry maintains, "If there is highly biologically productive land that is not used for farming ... it must be that it has other disadvantages in terms of transportation, land use regulation, or competition with alternative uses." He remarks that, if the TEM analysis is to be believed, there are actually several areas in which the average yield on currently uncultivated land would be higher than on cultivated land, which is economically counterintuitive.[72] On the other hand, Reilly (2010) contends that, because in developing regions such as Latin America land use is limited by infrastructure, it is likely that parcels of high-yielding land exist, and so GTAP should allow marginal to average yield ratios to exceed values of 1. At this stage, without additional work to demonstrate real correlations between the TEM model results and observed yield, it remains unclear whether this approach is a genuine improvement over a flat ratio assumption or whether it simply introduces an additional layer of uncertainty.

69 Tyner et al. (2010, p. 58) describe this as a "generic C4 crop." It has been confirmed via a personal communication that this was corn.

70 According to John Reilly's comments to the Air Resources Board (2010).

71 There is evidence that the harvest index (the ratio of yield to total biomass, or NPP) can be reduced under stressful conditions (reviewed in Hay 1995), which are more likely to be experienced by crops growing on low-quality land.

72 Recognizing that a marginal to average ratio greater than one seems unlikely, Tyner et al. (2010) cap the ratio at 1.

The question of marginal to average yield ratio is clearly an important one, but there is limited consensus in the literature as to what this ratio should be: estimates range from 0.29 (Edwards, Mulligan, and Marelli 2010, p. 103)[73] to above one for some regions based on the results from TEM. Assuming that the range identified by Keeney (2010) of 0.5–0.9 is probably reasonable, the effect on ILUC estimates could be substantial if the appropriate values are toward the lower end of that range.

3.2.6. Yield in the modeling of indirect land use change

Questions about yield, and especially about how and whether price increases caused by biofuel demand will induce general yield increases, have been at the heart of the ILUC modeling discussion from the start. Searchinger et al. (2008, p. 2), as already discussed, made the assumption that "present growth trends in yields continue but that positive and negative effects on yields from biofuels balance out." This assumption is clearly a substantial simplification of a number of complex interacting processes, and more recent modeling of ILUC has sought to treat various yield effects explicitly.

Hertel et al. (2010a), modeling with GTAP, included both a price-induced yield increase and a reduced yield at the margin of production. The yield on price elasticity in the modeling was set at 0.25, based (as noted above) upon Keeney and Hertel (2008). The yield on newly cultivated land was fixed at 66 percent of the average yield for each region, while the yield when one crop type expands onto land previously used for a different crop was based on the land rent (assuming that land with a higher rent will generally be more fertile).

Hertel and colleagues (2010b, p. 228, Table S1) broke down their results, allowing others to identify to some extent the impact of intensive and extensive yield changes. Within the United States, price-induced yield increase lessens land demand by about 18 percent, while lower yield at the margin of cultivation boosts land demand by 22 percent. Within the United States, yields of crops other than corn, such as oilseeds, tend to fall—presumably, as corn prices increase, other crops are pushed into lower-productivity cropland and onto virgin land. This reflects the knock-on yield impacts discussed by the Joint Research Centre (Edwards, Mulligan, and Marelli 2010).

For the rest of the world the picture is somewhat different, with a small overall land saving owing to yield effects. Apparently, as land switches between crops, the average yields for all categories increase, and this is more important than yield losses as new land is cropped. The Hertel model details do not indicate the land saved globally by yield effects, but the Joint Research Centre (Edwards, Mulligan, and Marelli 2010) provides a global breakdown of results for what is essentially the same model and scenarios, using GTAP to simulate an increase of 1 megaton of oil equivalent in U.S. corn demand. It finds that intensive yield effects provide 14 percent of the feedstock needed for corn ethanol. If the land 'saved' is distributed evenly around the areas where GTAP predicts expansion, this represents a 20 gCO_2e/MJ reduction in carbon intensity for corn ethanol.

The Joint Research Centre (Edwards, Mulligan, and Marelli 2010) results for the land area saved by price-induced yield increase show a wide variation between models (Figure 3.5). The modeling using IFPRI's IMPACT, for instance, has extremely strong yield effects, whereas the effects in GTAP are more modest but still important.

73 This assumes (for wheat) that land for expansion will have 0.7 times the average yield on a given farm and that the crop that expands will be rye with a yield 0.41 times the average for all cereals. Noting (Keeney 2010) that U.S. studies as well as the U.K. Department of Environment, Food and Rural Affairs (U.K. DEFRA 2001) found much less striking on-farm discrepancies, this suggestion may be reasonably treated as an outlier.

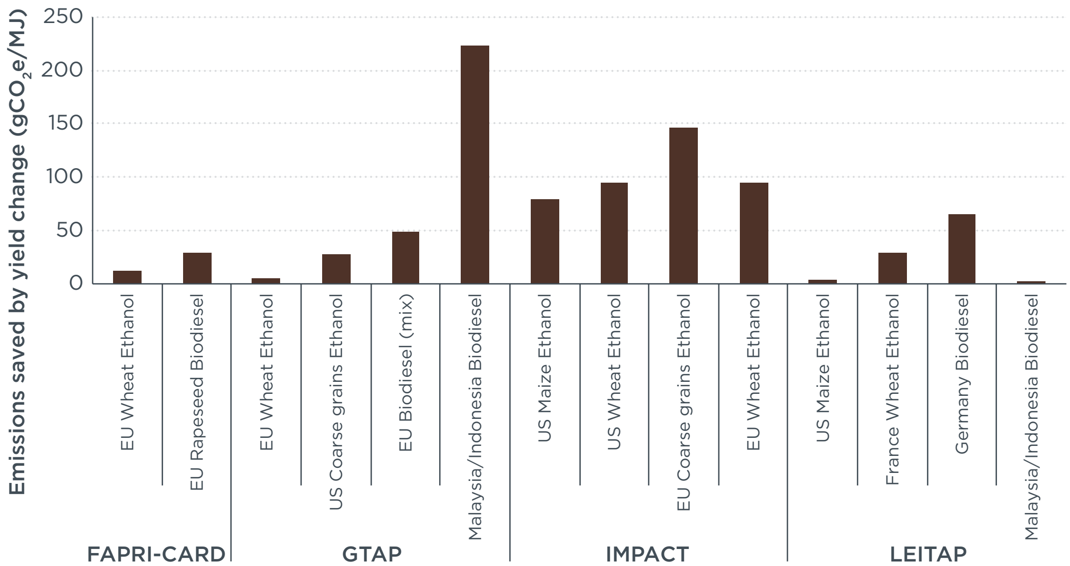

Figure 3.5 Comparison of the importance of price-induced yield change in models

Source: Edwards, Mulligan, and Marelli, 2010

3.2.6.b. Effect of marginal yield in the economic modeling

ILUC modeling with both GTAP and MIRAGE has tended to assume that marginal yields are lower than average yields. Keeney (2010) presents results from studying the sensitivity of land use change in Brazil predicted by GTAP, using the range previously given, from 0.5 to 0.9, for the ratio of marginal to average yield. He finds that the predicted extent of land use change responds to the marginal to average yield ratio exactly as might be expected—if the ratio is 0.5, 1.8 times as much land is needed as if the ratio were 0.9. Presumably, in any given region ILUC emissions predicted by GTAP would match the scale of land use requirements, so this modeling suggests that ILUC emissions will be sensitive to the marginal to average yield ratio, with the effect directly proportional to the chosen value. If this were carried through in the Tyner et al. (2010) modeling using TEM, which found ILUC emissions intensity for corn ethanol of 18 gCO_2e/MJ, then ignoring marginal yield would likely reduce emissions by about 2 gCO_2e/MJ, while using the original factor of 0.66 might increase emissions by about 7 gCO_2e/MJ (about 35 percent). Hertel et al. (2010b) analyze[74] the combined effect of both types of extensive yield modeling (the yield on new land and the yield as crops shift) and get a result that seems consistent with the sensitivity analysis by Keeney, with extensive effects increasing ILUC emissions by about 50 percent. The naïve expectation would have been for the yield effect on new land to increase emissions by about this much, which suggests that the effect of crop switching is less pronounced.

Laborde (2011a) also provides sensitivity analysis to the chosen value of the marginal to average ratio but finds a very different result. Unlike Keeney's modeling, where the results are sensitive to the parameter, Laborde notes that in Monte Carlo analysis with MIRAGE, the response to this parameter is ambiguous (Laborde 2011a, p. 32). Part of the

74 This analysis is done by switching effects on one at a time. Because the extensive effects are the last "switch," their importance ought to be well captured by this method (because as the last switch is turned on, all of the other possible economic responses are already active).

difference in responses between GTAP and MIRAGE is because MIRAGE has a dynamic baseline projected forward to 2020. That means that if the marginal to average yield ratio is low more expansion of area cultivated occurs in the model baseline, before the scenario with heightened biofuels demand is even run. Laborde explains, "The large expansion in the baseline needed to compensate for the low productivity of new land reduces the remaining amount of available land in the baseline." Ultimately, with the new baseline the model is more resistant to land expansion because of increased biofuel demand (so demand and price-induced yield effects are stronger in comparison). The upshot is that the effect of marginal yield on emissions may be limited in MIRAGE modeling. Laborde comments that this unexpected dynamic "emphasizes the role of the baseline behavior in our assessment and the importance of understanding that we compute the effects of the biofuel policy as a marginal deviation from this baseline."

3.3. CHOICE OF CROPS

The average yield in tons per hectare can be altered without changing the average yield of any given crop. Price induced yield change, as discussed in section 3.2.2, is about improving the yield of a given crop on a given piece of land. An economic model might predict, say, that the yield of wheat in France would rise by 10 percent in response to financial pressure. However, the total productivity of the system can also be affected by changing which crops are grown and where they are grown. In the words of the Joint Research Centre (Edwards, Mulligan, and Marelli 2010, p. 24), "average yields also go up or down because of changes in the crop-mix…, and because of production of individual crops shifting from one country to another."

3.3.1. Crop switching

Consider the example of corn and soybeans grown in the United States and Argentina. According to the FAO, the two countries have similar soybean yields, about 9 metric tons per hectare, but the U.S. corn yield is higher than Argentina's. If all 3 megahectares of Argentine corn-growing land were shifted to soy (assuming for a moment that it could attain average soy yields), this could free up 2.9 megahectares from growing soy in the United States, which could be changed over to corn. Overall, soy production would have remained the same, while total corn production in the two countries would have increased by 1.5 percent. It is easy to imagine that such crop switching between regions might be able to provide substantial efficiency in land productivity when considered for all crops and regions.

The example above made sure to preserve the overall production of soy. A more disruptive form of crop switching can occur if a shift to a crop with a higher yield occurs at the expense of production of a lower-yielding crop. For instance, ILUC modeling using IFPRI's MIRAGE model by Laborde (2011a) predicts a substantial reduction in the area in Europe devoted to 'other oilseeds' (an aggregated category consisting of various minor oilseed crops), with a simultaneous increase in rapeseed area. In MIRAGE, rapeseed is characterized as having a yield ten times the average for the other-oilseeds aggregate. Therefore, if rapeseed is allowed to displace other oilseeds in the model, the productivity of that land area in terms of tons of oil production per hectare can be substantially increased —but at the expense of reducing overall production of those other oilseed crops.

In real life, one would expect to see some limitations on the crop-switching processes just described. First, crops are not perfectly interchangeable, and a good area for corn production might not be such a good area for soy production. The example in which soy production shifts from America to Argentina relies on being able to achieve decent soy yields on the Argentine areas currently occupied by corn and decent corn yields on some of the U.S. areas currently used for soy. If it is impossible to achieve average or close to average yields after these shifts, then 1.5 percent would be a significant overstatement of the potential to increase corn productivity by a simple land swap of this sort. Following the logic of the Joint Research Centre (Edwards, Mulligan, and Marelli 2010; see section 3.2.5 above) it might not be appropriate to assume that the same yields could be achieved following crop switching as on existing areas since farmers are likely to have already optimized their land allocations. In general, ILUC models allow these types of area changes without applying any yield penalty. Where crop switching is an important contributor to meeting the demand for biofuels, this could cause ILUC to be underestimated, maybe substantially.

3.3.2. Location of expansion

In modeling ILUC, the location of crop expansion is critical because of the different emissions factors for different regions and land cover types (e.g. forest typically sequesters more carbon than grassland). Hence, if soy expansion in Brazil is believed to be linked to Amazon deforestation, then this may be more of a concern for ILUC than soy expansion in the United States.

The location in which expansion occurs is important not only for emissions factors but also in terms of determining the net land expansion because of the variation in yield between regions. The example from the previous subsection indicated that the United States has higher corn yields than Argentina (indeed, the United States has the world's highest, a consequence of many years of the development of a heavily industrialized system). When corn demand increases in the United States due to corn ethanol requirements, the acreage dedicated to corn production is expected to expand somewhere (i.e., some ILUC will likely occur, as well as reduced consumption and increased yields).

In calculating the quantity of land use change that this implies, it is important to determine whether the expansion of corn cultivation happens in the United States, at very high yields, or in a country like Argentina, with middling yields, or in a region like Africa, where, according to the FAO, the average corn yield is less than 25 percent of the U.S. yield. That means that it could require four times as much land use change in Africa as in the United States to meet the same increment in corn demand, with correspondingly larger carbon emissions. When the additional biofuel demand is coming from within the United States, this becomes a question of how fluid international trade is and how interconnected markets are —does an increase in demand in the United States translate to higher corn prices and additional production in Africa or anywhere else?

Economic modeling of ILUC recognizes the importance of this question, and models in general adopt one of two approaches to world trade. The first is to assume that there is a single world market for agricultural products. In this type of model, goods produced in all regions form a sort of indistinguishable central pool of available products, and consumers in all regions can purchase from this global pool without reference to where goods were produced. Increased demand in any region affects a world market price, and this world price applies to all producing nations. The approach can be made more sophisticated and realistic by including a more complex model of price transmission from the world market to local markets—for instance, by building in tariffs and transport costs. A single world market perspective deems that an African farmer will experience more or less the same change in demand as an American farmer or an Argentine farmer. This is the approach adopted by models including FAPRI, the model used by the EPA to assess ILUC. Models that take it up tend to show land use change being spread across a wider set of countries, with less focus on the country in which the biofuel mandate is originally introduced.

The other modeling approach uses what are referred to as 'Armington elasticities' to characterize trade relationships between each pair of countries. This approach is based on a 1969 paper, "A Theory of Demand for Products Distinguished by Place of Production," by Paul Armington, an economist working at the International Monetary Fund. According to Armington (1969, p. 1), "In theories of demand for tradable goods, it is frequently assumed that merchandise of a given kind supplied by sellers in one country is a perfect substitute for merchandise of the same kind supplied by any other country.... A preferable approach would be to recognize explicitly that any world model of feasible dimensions would identify few, if any, kinds of merchandise for which the perfect-substitutability assumption is tenable."'

That paper changed the way that people approach computable general equilibrium modeling. The idea of 'perfect substitutability' that Armington challenged is the single world market version of the world. As an alternative, he proposed that consumers would treat differently more or less identical goods that were produced in different regions. Consider a simple example of this type of differentiation in the automotive market: People might expect Japanese cars to be more reliable than Russian ones. In such a market, it might be too much of a simplification to say that Russian cars are perfectly substitutable for Japanese cars. There may be several reasons underlying the differentiated preferences that Armington describes, going beyond simple consumer preference to include transport costs, corruption, import tariffs, other import regulations, and imperfect information flows. To simplify all of this into a form that can be used in economic modeling, Armington collects all of these constraints into one number, the Armington elasticity between products from different regions.

The use of Armington elasticities can have a profound effect on the outcomes of indirect land use change modeling. For instance, when modeling increased demand for corn ethanol in the United States, a low elasticity between domestic and imported corn would tend to keep land use change effects concentrated domestically. The Joint Research Centre (Edwards, Mulligan, and Marelli 2010, p. 109) explains, "The results of GE models like GTAP and LEITAP show a strong correlation between the regions where ILUC occurs and the regions where extra biofuel production takes place. This is a result of the Armington elasticities used in GTAP and LEITAP."

Keeping in mind that the United States has the highest corn yields in the world and Africa has low ones, a model with a low Armington elasticity between American and African corn would be likely to show a lower net land use change than one that assumed a single world market.

One problem with using Armington elasticities is that they are derived from existing trade flows, and that makes them 'sticky' in the model—that is, the model may be unable to reflect fundamental changes in global trade patterns. For instance, imagine that in the next 10 years an African government strongly encourages export-oriented agricultural development, specifically with a view to selling to biofuel markets. If that country does not currently have substantial trade relationships in the commodity in question, in any economic modeling the Armington elasticities will prevent them from developing in future. To take another example, Armington elasticities for purchases of Japanese cars in Europe based on data from the 1950s would be a poor picture of those purchases today.

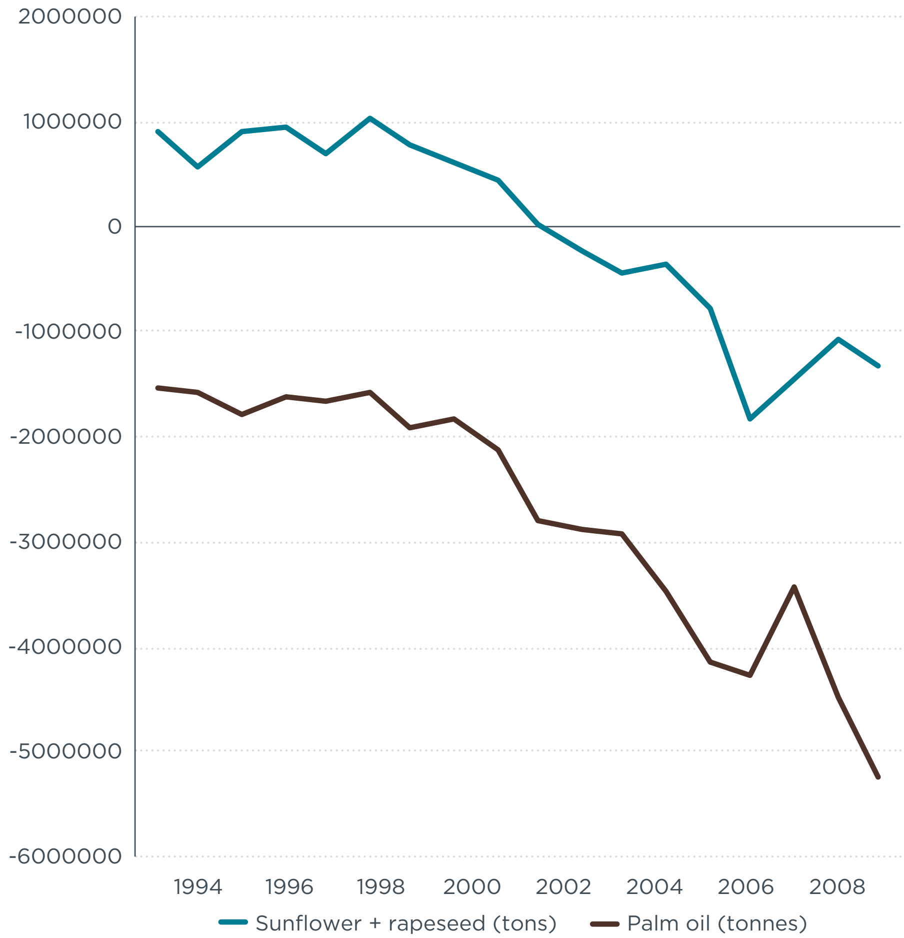

Figure 3.6. European trade balance for palm oil and the aggregate of sunflower oil and rapeseed oil

Source: FAOSTAT, via Malins, 2011a

Note: Net exports are positive, net imports negative.

The use of the Armington framework has been the subject of intense debate in economic modeling, both for ILUC and for other applications. The Joint Research Centre (Edwards, Mulligan, and Marelli 2010) notes that Armington elasticities are generally calibrated using short-term data. While there may be some inertia in the system in the short term, in the long term markets can evidence considerable flexibility. For example, a single poor rapeseed harvest might result in stocks being drawn down, with a relatively weak transfer of demand to other oils, but a sustained increase in the rapeseed oil price could drive users to find ways to use other oils with slightly different properties. The JRC (Edwards, Mulligan, and Marelli 2010, p. 109) therefore contends that Armington-using models are likely to over-concentrate land use change within the area in which mandates occur. An example would be the GTAP modeling for the state of California, in which land use and consumption changes occur to a much larger extent in the United States than in the single-world-market-based FAPRI modeling.

3.3.2.b Vegetable oil markets

For indirect land use change modeling of biodiesel in particular, one important question is the extent to which different oils can substitute for each other. Specifically, because of the strong link between palm oil production and deforestation, the amount of palm oil expansion predicted by a given model can be critical. At one extreme, E4tech's causal descriptive modeling assumes that other oils cannot substitute for rapeseed oil at all in European food markets (E4tech 2010, p. 52), "While significantly cheaper than rapeseed oil, palm oil use tends to be constrained due to its physical properties (principally the fact it is solid at room temperature). Experts tend to believe that palm oil has already achieved its technical maximum market share in the EU (particularly EU15 countries)." In Armington terms, this assumption would mean a very low Armington elasticity between European oilseeds and those from Indonesia and Malaysia.

At the other end of the spectrum, several analysts argue that over the medium to long term vegetable oils are completely substitutable. It might take some time for people to learn to accommodate different products, but, given a prolonged price signal, they would happily switch. Here is the Joint Research Centre (Edwards, Mulligan, and Marelli 2010, p. 109) again, "A rapeseed oil 'shortage' which lasts a decade because of an increase in demand due to biofuels will result in much more widespread effects, as manufacturers and consumers learn to use substitute oils, and as production area expands in regions with more available land."

Evidence that European rapeseed oil demand is well correlated to demand for other oils, notably palm oil, is laid out by the Malins (2011a; 2013) based on FAO data. Figure 3.6, shows that the overall European trade balance of domestically produced oils, sunflower and rapeseed, has tracked closely the trade balance in palm oil, suggesting that demand for domestic oils is well correlated to demand for palm. Malins (2013) also notes that growth in European vegetable oil imports has been coincident with growth in biodiesel demand.

3.3.3. Crop location and switching in the modeling of indirect land use change

Searchinger et al. (2008, p. 2) stress the importance of the location of crop expansion to determining land use change effects:

- As more American croplands support ethanol, U.S. agricultural exports decline sharply (corn by 62%, wheat by 31%, soybeans by 28%, pork by 18% and chicken by 12%).
- When other countries replace U.S. exports, farmers must generally cultivate more land per ton of crop because of lower yields.

The extent to which ILUC will occur within or outside the United States has been much debated. Searchinger and colleagues, using the FAPRI model which has a single world market view of agriculture, find very large reductions in exports; hence, expansion of cultivated land is predicted elsewhere in the world. GTAP, espousing the Armington assumption that tends to keep more production based domestically, finds ILUC results focused more within than beyond the United States.

Crop switching has not been one of the more thoroughly discussed ILUC effects. Because, even more so than for changes in food consumption or yield, it is an expression of the interaction of a long list of elasticities representing different crops, markets, and countries, it can be difficult to get a handle on. The decomposition methodology proposed by Witzke et al. (2010), with some modifications (see Appendix A), can highlight the effect of area shifting between regions in various scenarios (Figure 3.7).

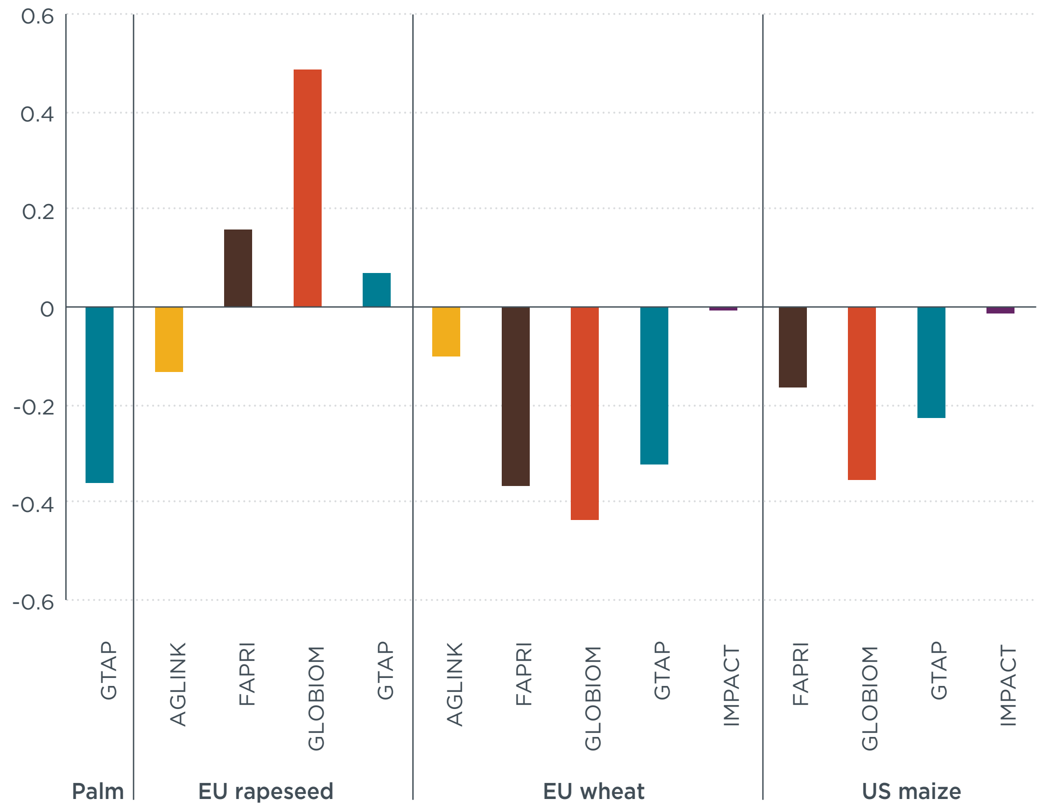

Figure 3.7. Effect on land use of expanding crops in areas with yields higher or lower than the world average based on methodology outlined in Witzke et al. (2010)

Note: Positive values represent land required for expansion of cultivation; negative values represent land conserved.

Notably, the savings or increases in land requirement implied by shifting the location of production are of the same order of magnitude as the savings identified by the Joint Research Centre (Edwards, Mulligan, and Marelli 2010) from price-induced yield effects. Also, despite their different treatment of agricultural markets, both GTAP and FAPRI (in this cases a later version than that used by Searchinger et al.) concentrate additional production in higher-yield areas. There is no question that the model decisions about where crops will be expanded are a key component of the ILUC calculation.

3.4 UTILIZATION OF CO-PRODUCTS

Biofuel production does not fully utilize the crops used as feedstocks. Traditional ethanol production uses fermentation of carbohydrates, so that proteins and fats in the feedstock are left over. Oilseed crushing produces oil for biodiesel, but there are proteins and carbohydrates left over in the meal.[75] Biofuels production is therefore accompanied by the production of economically valuable co-products, notably distillers grains with solubles (DGS) and oilseed meals. In the United States, corn DGS and soybean meal are the two major co-products and are both used as animal feed. In the EU, wheat DGS and oilseed rape (OSR) meal are the major co-products. As in the United States, these co-products will find increasing use in animal feed rations in the EU. They are important

75 A meal is the powdery substance produced by crushing a grain, seed, or pulse.

sources of protein and energy and can readily be integrated into rations for cattle, pigs, and poultry. Table 1 shows some examples of co-products produced from biofuel pathways and their potential economic uses.

Table 3.2. Some examples of co-products from biofuel production

Biofuel	Co-products	USES
Corn Ethanol	Dry milling—Wet and dried distillers grains with solubles (DGS) Wet milling—corn gluten meal, corn gluten feed, corn germ meal, corn oil, corn steep liquor	DGS in animal feed—displaces corn, urea, soybean meal, etc. Corn gluten meal, corn gluten feed, corn germ meal, corn steep liquor—can be used for feed rations of various livestock types Corn oil—biodiesel, cooking oil, pharmaceuticals
Wheat ethanol	Wheat dried DGS	Displaces wheat, barley, soybean meal, etc., in animal feed
Biodiesel from vegetable oils (soybean, rapeseed, sunflower, palm oil, etc.)	Meal Glycerol	Meal—source of protein in animal feed Glycerol—pharmaceuticals, chemicals, cosmetics, energy
Cellulosic biofuel	Electricity from lignin	Displaces grid electricity
Sugarcane ethanol	Electricity from bagasse	Displaces grid electricity

Co-products can be broadly placed into two categories: those that directly displace land-based products and have land use implications, such as DGS displacing soybean meal, and those that displace non-land-based products such as urea, glycerol, and electricity. Co-products in the second category do not have land use implications but have greenhouse gas (GHG) reduction implications.

When biofuel co-products are utilized as livestock feed, this reduces demand for other commodities and the associated demand for land. It is important to consider this in any assessment of indirect land use change. The corn ethanol industry uses about 40 percent of the U.S. corn crop, but about a third of this is returned to the livestock feed market as distillers grains, potentially reducing the net increase in land demand by a third as well.

As cellulosic ethanol production scales up, co-product electricity from lignin is likely to be used to displace grid electricity. This probably does not affect land use but may provide an emissions credit, which is often counted in the direct life cycle analysis (LCA). Likewise, electricity generated from bagasse is a co-product in sugarcane ethanol production. Although it is important to account for reductions in GHG emissions from displacement of non-land-based products in estimating well-to wheel GHG emissions, the focus here is on the co-products with land use implications, as they are most relevant to ILUC modeling.

3.4.1. How do co-products affect land use?

In general, the land use impact of co-products depends on

» Co-product yield per unit biofuel

» The amount of other feedstock that is displaced (i.e., does one ton of co-product replace less than one ton, one ton, or more than one ton of other feed?)

» The variety of other feedstock that is displaced—does the co-product replace only corn, or both soybeans and corn, or a combination of soybeans and corn and alfalfa and wheat middling (the portion of the kernel not typically used as grain) and almond hulls, etc.

» Land use requirements of all displaced feed components.

To illustrate how co-products can affect land use emissions, consider the example of wheat dried distillers grains (DDGS), an important co-product in wheat ethanol production in Europe. As the demand for ethanol grows, the output of wheat DDGS increases. Wheat DDGS serves as an animal feed resource and can replace wheat, soybean meal, and other ingredients—the displacement ratio for each of these ingredients is essential to know. Assuming that reduced feedstock demand translates to lower land demand, whatever amount of cropland would have produced the animal feed now displaced will no longer need to be cultivated. Either this land becomes abandoned or further expansion into wild land is curtailed.

Not all of the knock-on effects will cut back on land use, though. Say that wheat dried distillers grains displace some soy meal, reducing soybean production. This should result in less cropland expansion in countries like Argentina and Brazil. However, at the same time as production of soybeans for meal dwindles, so does production of soybean oil, and the demand for soybean oil then has to be met from other vegetable oils such as palm oil from Indonesia and Malaysia. This could lead to the growth of oil palm plantations in Indonesia and Malaysia, increasing GHG emissions.

This brings up the question of what the carbon implications are of all of these knock-on effects and in particular whether the carbon ramifications of expanding wheat in Europe in the first place are greater or lesser than the consequent cultivated land shrinkage in South America and land expansion in Southeast Asia. It has been argued (Morton et al. 2008) that soybean expansion in South America is well correlated with rain forest destruction. If that is true, then the carbon credit from avoiding soy expansion may well be significant compared with the emissions from wheat expansion. Soy also has a lower per hectare yield than wheat, so the land area saved by displacing soy could also be large compared with the wheat expansion area.

On the other hand, it is generally accepted that palm oil expansion is even more strongly correlated with rain forest destruction than is soy expansion. More than that, in Indonesia and Malaysia, at least one-third of oil palm expansion in the coming decade is expected to occur on peatland, with enormous associated carbon emissions (Mietinnen et al. 2012). Therefore, although the increase in the oil palm cultivated area will be small compared with the potential expansion in soy acreage averted, it could represent an important deficit, pushing wheat ILUC back up again.

This narrative tends to assume a fairly direct transmission of information in the market—one feedstock displaces another, which in turn displaces another, and so on. In MIRAGE modeling by Laborde (2011a), one can see the importance of subtler price effects on the ILUC credit or deficit accruing to co-products. Malins (2011a, p. 5) has outlined such dynamics in play, “The combination of increased wheat/maize demand for ethanol, and increased supply of protein feed in the form of DDGS, is modeled to alter the balance of energy and protein feed use across the livestock industry. Reduced protein feed prices drive an overall increase in protein feed demand, making up for the soy meal etc. that was displaced by distillers grains so that there is very little net change in the consumption of soy meal in these scenarios.”

In Laborde’s modeling, even though it is assumed that DDGS displaces soy meal directly, overall soy meal consumption in the livestock sector barely changes. If the model economics are a good reflection of real-world economics, this suggests that caution should be exercised about assumptions that co-product effects are a function solely of the feed products that will be directly displaced. In general, while the expectation that the net impact of co-products will be to reduce ILUC emissions is reasonable, there can be complex offsetting market effects, and a simple model of feedstock displacement might not fully capture the effects.

The results from agroeconomic models confirm the significant land use impact of co-products. For example, GTAP modeling for wheat ethanol, oilseed biodiesel, corn ethanol, and palm oil biodiesel suggest that the corresponding co-products can reduce the net land use change by 30 percent, 52 percent, 46 percent, and 22 percent, respectively (Edwards, Mulligan and Marelli 2010). These results were obtained by rerunning the GTAP model without considering co-products.

3.4.2. Ethanol co-products

When ethanol is produced from cereal crops (notably corn and wheat), about a third of the digestible energy of the plant is not converted but remains in the co-products, generally referred to as distillers grains and solubles (DGS). These can be dried to produce 'dried distillers grains and solubles,' DDGS, or sold locally as 'wet distillers grains and solubles,' WDGS. Dried grains, DDGS, have received more attention from modelers and in feed trials, and thus results for DDGS have often been applied to DGS in general. Similarly, when sugar beet is processed into ethanol, beet pulp is left over. When ethanol is produced from sugarcane, fibrous bagasse is left over. This has more limited feed value than distillers' grains or beet pulp, but in Brazil it is normally used for biomass energy to power sugarcane-processing plants. The amount of co-product that is produced per metric ton of ethanol is relatively uncontroversial for each feedstock, but there is ongoing discussion about exactly what and how much these co-products displace and what this means in carbon terms.

In determining the impact of co-products used as animal feed on indirect land use change emissions, the key question is what land-intensive feedstock(s) the co-product will displace in the market. Co-product displacement ratios[76] figure prominently in indirect land use change modeling and life cycle analysis. Kim and Dale (2005) used a system expansion[77] approach to account for GHG impact of co-products using displacement ratios. Lywood, Pinkney, and Cockerill (2009) calculated the amount of land saved if wheat and soybean meal were displaced by wheat DDGS according to energy and protein content and took that figure to estimate the land use impact of wheat bioethanol production. Their calculation is based on the assumption (probably reasonable, up to a point) that expansion of soy production will have a larger carbon footprint (notably through deforestation) than expansion of wheat production. All major ILUC models now include systems to account for the disposition of co-products.

In most of the existing literature on this topic, displacement ratios have been determined in one of two ways, either by feed trials or by comparing nutritional content. The U.S. work on displacement ratios has been largely based on data from feed trials. For instance, a review conducted by the Hoffman and Baker (2011) showed that all previous experimental studies in the United States indicated that DDGS would displace mainly corn and soybean meal, and in some cases urea (which is used as a feed additive for ruminants) (Arora, Wu, and Wang 2008). The review found that on average 1 unit of

76 The amount of each competing feed product that a metric ton of co-product will displace.

77 In lifecycle analysis, system expansion involves analyzing what is replaced in the market by an increased supply of some co-product, and determining an emissions credit (or deficit) related to this replacement. It is an alternative to allocating the emissions from the main process between several co-products, based on mass, energy content, economic value or some other characteristic.

DDGS displaces 1.22 units of the total animal feed consisting of corn and soybeans in the United States. Table 3.3 shows the conservative and potential displacement rates by livestock and feed type identified from the literature.

Table 3.3. Corn DDGS displacement ratios reported in Hoffman and Baker (2011)

Cases		Livestock type	Corn	Soy meal	Total disp. ratio	Remark
Displacement rates estimated from literature	Conservative assumptions	Beef	1	0	1	Beef cattle data from Vander Pol et al. (2006); Trenkle (2003). Dairy cattle data from Anderson et al. (2006). Swine data from Shurson and Mindy (2002), Shurson et al. (2003). Poultry data from Lumpkins et al. (2004; 2005); Roberson (2003).
		Dairy	0.45	0.55	1	
		Swine	0.89	0.1	0.99	
		Poultry	0.51	0.5	1.01	
	Maximum substitution potential	Beef	1.2	0	1.2	Beef and dairy cattle data from Arora, Wu, and Wang (2008). Swine and poultry data from Shurson (2009a; 2009b).
		Dairy	0.73	0.63	1.36	
		Swine	0.7	0.3	1	
		Poultry	0.61	0.44	1.05	

The second approach, which has been more dominant in Europe, is to assume that co-products will exactly substitute for the nutritional value of the feed they replace. For instance, Lywood, Pinkney, and Cockerill (2009) assume that distillers grains have two basic nutritional components (energy and protein) and will displace a combination of wheat feed and soy meal. Having determined the respective energy and protein content of each of the three feed products, it becomes a trivial question of solving a pair of simultaneous equations to work out how much wheat and soy respectively must be replaced to maintain the overall energy and protein content of a diet.

The feed trial studies reviewed by Hoffman and Baker are all based on simplifications of animal feeding operations, which do not fully reflect the market realities. The initial purpose of the feed trials is not to judge which feedstocks are likely to be displaced by DDGS in the market but to assess the maximal inclusion rates of DDGS in animal diets without reducing growth rates or harming animal welfare. The trials usually measure diets with DDGS against simple comparison diets in order to give clear results; however, these simple trial diets are a poor model for real animal diets.

Hoffman and Baker suggest that 1 metric ton of DDGS will on average displace 1.22 tons of other ingredients. If that is correct, it should mean that accounting for co-products yields an increased ILUC ‘credit.’ However, the co-product subgroup report of the California Air Resource Board’s Expert Workgroup on Indirect Land Use Change (ARB 2010, p. 31) questions the validity of displacement ratios greater than one. It maintains that only if the baseline ration were suboptimal, as in the over-simplified baseline trial diets, would DDGS use improve animal performance, hence giving a displacement ratio larger than 1:1. When considering actual commercial animal diets, this is unlikely to be possible.

The nutritional-content-based work of Lywood, Pinkney, and Cockerill does not introduce the problem of a 'false baseline.' However, the assumption that only energy and protein content are nutritionally important is a simplification, as is the assumption that only wheat feed and soy meal are replaced in the feed market. In reality, commercial animal feed producers and farmers adjust their feed mixes to minimize cost using linear least-cost formulation computer models that incorporate detailed data about the nutritional content and current market price of each ingredient.[78]

Recognizing that the use of feed trial data may introduce systematic errors into displacement ratios assumed for co-products but that there are more determinants of livestock feed choices than simply energy and protein content, the ICCT has commissioned least-cost-formulation-based studies that aim to mimic closely the way feed choices are made by farmers. These studies (Hazzledine et al. 2011; Klasing 2012) suggest that co-products such as DDGS can displace a wide array of feed ingredients, not just corn and soybean meal.

Klasing (2012) models the U.S. animal feed market and finds that corn DDGS is likely to displace primarily corn feed but also a wide range of other feed ingredients (Table 3.4). There is relatively little soy displacement predicted. Crop-based products with direct land use implications account for 94 percent of net displacement, and the remaining 6 percent comes from ingredients without direct land use implications such as feather meal (from poultry feathers), calcium phosphate, and vitamin supplements.[79] Overall, 1 metric ton of DDGS displaces 1 ton of other feed.[80]

Table 3.4 DDGS displacement ratios in metric ton displaced per ton of DDGS for the major feed ingredients

Corn DDGS (Klasing 2012)		Wheat DDGS (Hazzledine et al. 2011)	
Corn dent yellow	0.55	**Soybean meal**	0.29
Soybean meal	0.07	**Barley**	0.30
Canola meal	0.06	**Sunflower meal**	0.18
Wheat	0.07	**Maize gluten feed**	0.13
Other wheat products	0.13	**Palm kernel extractions (PKE)**	0.12
Rice bran	0.05	**Wheat**	-0.06
Alfalfa	0.05	**Citrus pulp**	-0.03
Corn gluten feed	0.02		
Cottonseed meal	0.03		

Note: Obtained using least-cost formulations in the United States and the United Kingdom; minor ingredients like vitamins, calcium, feather meal, etc., are not shown. A negative displacement ratio means the use of DDGS requires addition of a given feed ingredient.

78 Other elements that must be balanced include sulfur (DGS can contain sulfur from sulfuric acid used in ethanol production) and phosphorus levels, and exposure to mycotoxins used to eliminate undesirable bacteria during ethanol fermentation must be monitored. Higher phosphorus levels in DDGS increase the amount of phosphorus in manure, thereby affecting its application rates.

79 Note that displacing production of these ingredients will still have GHG implications.

80 In some cases, considering only corn and soy displacement without reference to other ingredients could give highly misleading outcomes. For example, when adding 1 kg of DDGS to poultry diets is modeled to replace 2.2 kg of corn while requiring an additional 0.2 kg of soybeans—this would give a net displacement ratio of 2 if only these ingredients were considered. However, because the revised diet would in fact also require the addition of ingredients such as wheat flour middling and animal fats, the overall displacement ratio would come out as 1 after all.

Similarly, a study by Hazzledine et al. (2011) of markets in Great Britain shows that wheat DDGS is likely to displace a wide range of animal feed ingredients from rations (Table 3.4). Notice that, in the least-cost modeling, it is not necessarily just a question of displacing feed ingredients. In a complex diet, in some cases introducing distillers grains would actually cause farmers to use more of some ingredients. Thus, Hazzledine finds that use of wheat feed actually increases slightly for the scenario in Table 3.4. Nevertheless, the overall net displacement in each case comes out as one metric ton net reduction in other feed use for each one ton increase in distillers grains.

While the least-cost modeling represents a significant improvement over feed-trial-based results, there are many sources of uncertainty and variability that should be borne in mind. The Klasing (2012) modeling, for one, is grounded on feed ingredient prices at a single point in time, whereas the displacement expectations for some ingredients may be sensitive to shifts in their prices. Beyond this, the least-cost modeling detailed above cannot fully anticipate the knock-on price effects of displacement—there are feedback loops between prices that would require further economic modeling to consider.[81]

Overall, the simple premise that distillers grains will reduce net land demand in proportion to their weight may well be a reasonable approximation. For the U.S. corn ethanol market, this seems particularly true. The strongest case for bioethanol receiving a large ILUC credit due to co-product utilization is made by Lywood, Pinkney, and Cockerill (2009). Hazzledine et al. have shown that the strongest version of the assumption about soy meal displacement, as made by Lywood and colleagues and reflected in other European work like that of E4tech (2010), is probably not justified. However, there is certainly potential for a credit from this displacement effect that could justify a lower ILUC value for wheat ethanol. Even so, the latest MIRAGE modeling by Laborde (2011a) maps out a scenario in which, despite strong displacement of soy meal by DDGS, overall soy meal demand does not change substantially. Therefore, further examination of feed market dynamics would be advisable before making any strong assumptions that co-products should provide a major ILUC credit for wheat ethanol. Table 3.5 sums up the impact of ethanol co-products on land use.

81 Hazzledine et al. address this point somewhat by running alternative cases in which inclusion is fixed of ingredients that are likely to become cheaper in response to cheap DDGS supplies.

Table 3.5. How do ethanol co-products affect indirect land use change?

Key question	Assessment
Co-product yield per unit biofuel	Varies by feedstock type, but this is fairly well understood, with co-product yields defined for all major life cycle analysis models. Co-product yields are generally high enough to be an important ILUC consideration.
The amount of other feedstock that is displaced	The evidence is strong that, for biofuel co-products, one metric ton of co-product will displace about one ton of other animal feed products overall (although, within this dynamic, the use of some other products could actually increase; see Table 3.4).
The variety of other feedstock that is displaced	Distillers grains are high in protein compared to the base biofuel feedstocks (wheat or corn). Therefore, it is likely that they will displace other mid-protein or high-protein feed, as well as 'energy' feeds. The picture is more complicated, however, than simply replacing corn/wheat and soy meal. Also, Laborde (2011a) shows that the calculus can be more powerfully affected by the overall economics of the livestock feed sector as well as by simple nutritional concerns.
Land use requirements of all displaced feed components	It has been argued that soy expansion is more carbon intensive than wheat expansion. Certainly, corn and wheat have higher yields than soy. The best way to address this question is to let a land use model that has a good treatment of co-products solve it. For other displaced feeds (such as alfalfa or citrus pulp), there is a need for further consideration of the land use implications of displacement.

3.4.3. Biodiesel co-products

Biodiesel processing of oilseeds such as rapeseed, sunflower, and soybean produces meal during the crushing process. Meals are a cost-effective source of protein but also provide energy. Oilseed rape meal is less palatable to animals than soy; hence, it is mixed with other feed ingredients to improve palatability. As compared with soybean meal, OSR meal has lower energy and protein content. OSR meal has higher fiber content, which restricts its inclusion in non-ruminant diets. It also has higher total phosphorus, but its bioavailability (capacity for absorption) to monogastric animals is lower than for soybean meal. This suggests that if higher phosphorus content in manure is a concern, a higher inclusion rate of OSR meal could be problematic. Glycerol, another co-product of biodiesel production, can be used in the pharmaceutical and cosmetic industries.

As in the case of DDGS, displacement ratios of biodiesel co-products have been used in LCA and land use modeling (E4tech 2010; Lywood, Pinkney, and Cockerill 2009). The major biodiesel co-products for which displacement ratios could be important to ILUC modeling include soy meal, rapeseed meal, palm kernel oil (PKO),[82] and palm kernel extract (PKE).

Like distillers grains, meal largely enters livestock feed markets. Also like distillers grains, the displacement estimates in the literature are largely based on a simple comparison of metabolizable energy and digestible protein, with the assumption that only feed wheat and soy meal will be displaced. This methodology gives the displacement values in Table

82 E4tech (2010) argues that palm kernel oil will displace coconut oil on a 1:1 mass basis since the properties of palm kernel oil are similar to coconut oil. It expects that increased PKO availability will therefore reduce demand for coconuts and curtail land expansion. E4tech predicts that about 50 percent of oil palm expansion would be offset by reduced coconut-growing area in Indonesia because of relatively low coconut oil yields—if the relationship were that strong, one would, in the absence of other information, expect to see that oil palm area change was inversely correlated to coconut area change. Such a correlation is not observed in FAO data over the past 10 years of Indonesian oil palm expansion, suggesting that the E4tech hypothesis may not be correct, in which case PKO may instead have the net effect of slightly reducing palm area demand. Additional analysis of coconut oil/PKO demand dynamics would be necessary to properly confirm or refute it.

3.4, suggesting that rapeseed meal in general displaces more protein source (soy meal) than energy source (wheat), while PKE on the other hand displaces more energy source (wheat) than protein source (soy meal). Because these displacement ratios are derived from simplistic assumptions, they are unlikely to capture the full market reality. Table 3.6 shows the replacement rates found for an increased rapeseed meal scenario using the Hazzledine et al. (2011) least-cost feed formulation model. Especially if displacing soy is expected to provide the largest ILUC credit, it is of some importance to know whether meal displaces only 'core' feed ingredients (wheat and soy in Europe, corn and soy in the United States) or whether the market interaction is more complex.

Table 3.6. Reported EU displacement ratios for biodiesel co-products based on energy and protein content

Co-product	Livestock	Wheat	Soya meal	Total displacement ratios	Source
OSR meal	Swine	0.23	0.72	0.95	Croezen and Brouwer, 2008
OSR meal	Swine	0.18	0.56	0.74	Lywood, Pinkney, and Cockerill, 2009
OSR meal	Ruminants	0.28	0.60	0.88	Croezen and Brouwer, 2008
OSR meal	Ruminants	0.21	0.66	0.87	Lywood, Pinkney, and Cockerill, 2009
OSR meal	Ruminants	0.48	0.38	0.86	Marelli, Mulligan, and Edwards, 2011
OSR meal	Poultry	0.06	0.61	0.67	Lywood, Pinkney, and Cockerill, 2009
PKE	Swine	0.45	0.11	0.56	Lywood, Pinkney, and Cockerill, 2009
PKE	Ruminants	0.81	0.16	0.97	Lywood, Pinkney, and Cockerill, 2009

Table 3.7. Possible displacement ratios for rapeseed meal in the United Kingdom

Ingredient	Displacement ratio
Cereals	0.36
Maize gluten feed	0.32
Extracted soya bean meal	0.30
Extracted sunflower meal	0.13
Palm kernel extract	0.16
Fat	-0.05
Citrus pulp	-0.22

Note: Based on least-cost modeling

The main question with respect to soybean co-products is the extent to which soybean oil demand (for biodiesel) actually affects soybean meal output. Does an increase in soy oil demand lead to an increase in soy-growing area, or is vegetable oil demand fungible, so that increased soy oil demand increases the planting area for oilseeds generally? At one extreme, each additional 0.2 kg of soybean oil demand could result in 1 kg of new soybean production and thus 0.8 kg of extra soybean meal on the market. At the other extreme, soybean meal production might be unaffected by the consumption of soybean oil by the biodiesel market, with the primary effect being that additional oil is produced elsewhere to replace the soybean oil. The actual market behavior will be determined by the relative elasticities of supply and demand and should end up somewhere between these extremes.

Table 3.8. How do biodiesel co-products affect indirect land use change?

Key question	Assessment
Co-product yield per unit biofuel	As with ethanol, varies by feedstock type, but this is fairly well understood, with co-product yields defined for all major LCA models. Co-product yields are generally high enough to be an important ILUC consideration.
The amount of other feedstock that is displaced	Again, as with ethanol, the evidence is strong that for biofuel co-products displacing animal feed, one metric ton of co-product will displace about one ton of other products overall (although, within this dynamic, the use of some other products could actually increase; see Table 3.4).
The variety of other feedstock that is displaced	Oil meals are high-protein feeds and, like distillers grains, will displace other medium-protein or high-protein feed, as well as 'energy' feeds. Laborde (2011a) shows that the picture can be affected by the overall economics of the livestock feed sector as well as by simple nutritional concerns.
Land use requirements of all displaced feed components	As with ethanol, the best way to address this question is to let a land use model that has a good treatment of co-products and livestock feed markets solve it. The same argument has been made for rapeseed meal as for wheat DDGS (that it will displace carbon-intensive soy production). ILUC modeling should be able to assess the importance of this effect.

3.4.4. Co-products in the modeling of indirect land use change

Unlike traditional LCA and causal descriptive modeling, which directly use co-product displacement ratios for various feed ingredients to estimate the amount of land saved and hence GHG emissions reduced or avoided, agroeconomic models employ elasticities of substitution between co-products, protein, and energy feed.

The treatment of the displacement effects of DGS have generally been relatively simple in agroeconomic ILUC models, with the earlier models tending to ignore co-products altogether or else assuming that DGS displaces only whole corn (e.g., Tokgoz et al. 2007) or whole corn and soybean meal (e.g., Hertel et al. 2010a). The modified GTAP model by Tyner et al. (2010, p. 15) captures more substitutions between animal feed ingredients and DDGS by creating a three-level nested structure. This allows for substitutions at the first level between distillers grains and cereals and between oilseed meals and oilseeds. At the second level, it allows for a substitution between the sum of oilseeds and oilseed meals (protein feed) and the sum of distillers grains and cereals (energy feed). This composite of all four basic feedstuffs can then be substituted at the highest level of the nest by other crops, processed feeds, and livestock products. In this structure, it would be possible in theory for increased oilseed meal supply to have the effect of substituting for some sugarcane at the top level of the nest, but in practice it is likely that the substitution will be strongest at the lowest level.

IFPRI-MIRAGE (Laborde, 2011a, p. 106) now includes a modeling of the livestock feed sector that is not only based on elasticities of substitution but also contains information about the energy and protein content of different feeds. This means that high-protein co-products preferentially displace other high-protein feedstuffs such as soy meal. However, Malins (2011a, p. 5) observes that, while the initial substitution is primarily for other high-protein feeds, Laborde finds that (because of additional price effects) the net effect of increasing the supply of corn distillers grains is that demand for corn feed falls. This is presumably because the increase in availability of protein feeds makes them cheaper and encourages a general shift toward feeding animals more protein.

The Joint Research Centre (Edwards, Mulligan, and Marelli 2010, p. 90) compares the effect of co-products on ILUC estimates across several models (Figure 3.8). IMPACT did not model co-products at all, while the results for LEITAP seem implausibly low. For the FAPRI and GTAP modeling, which do make a serious attempt to consider co-products, JRC found a major contribution toward reducing net land use requirements—up to 60 percent in the case of rapeseed. The results for ethanol are similar between FAPRI and GTAP, for wheat and corn, which reflects the tendency to assume in these models that distillers grains would replace corn/wheat feed in proportion to the mass produced.

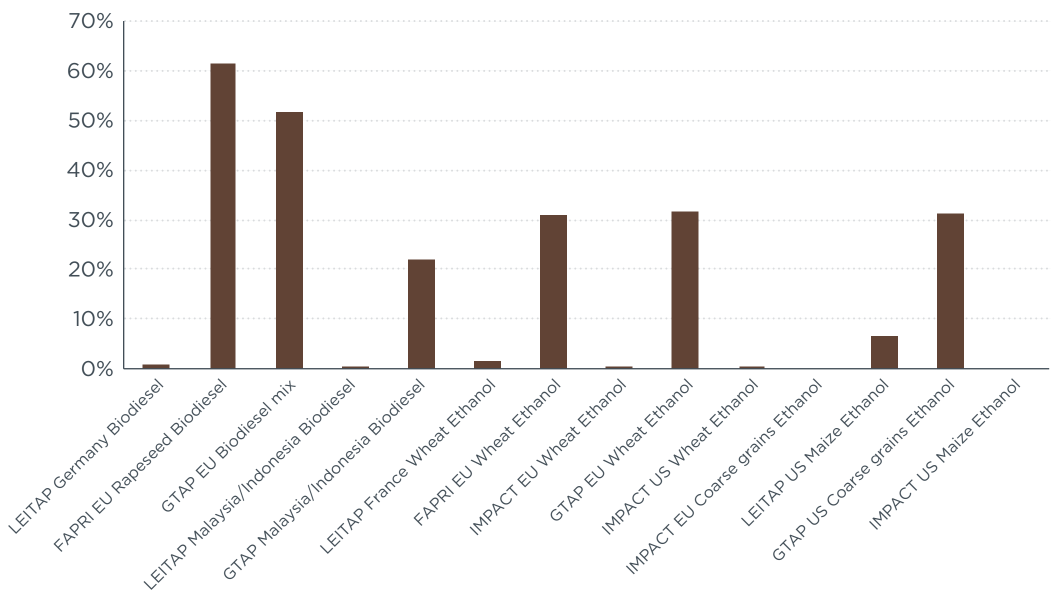

Figure 3.8. Percentage by which the availability of co-products reduces overall ILUC in various modeled scenarios

Source: Edwards, Mulligan, and Marelli, 2010

3.5. ELASTICITY OF AREA TO PRICE

So far, this chapter has discussed one major 'demand side' response to the increased feedstock prices that come with heightened biofuel demand—food consumption reduction—and one major 'supply side' response—price-induced yield increase. It has also talked about related developments that can reduce or increase the net land demand for biofuels—co-products and crop switching. Of course, this leaves one further supply-side response (which has already been mentioned several times), the most important for discussing indirect land use change—the area response.

The area response to increased demand is not just a remainder term, determined by the difference between gross feedstock demand for biofuels and the amount of feedstock supplied by yield increase and consumption reduction. The area response will happen simultaneously with any yield and demand response. By limiting price increases, each of these responses affects the magnitude of the others. As the Joint Research Centre (Edwards, Mulligan, and Marelli 2010, p. 12) notes, "All models are sensitive to the ratio of yield to area elasticity in different countries." One of the most important questions with respect to modeling ILUC is therefore how strong the area response to price increase will be and how it compares with other responses?

What evidence is there for the strength of the area response, or indeed for the proposition that the area under cultivation responds to price at all? The basic economic argument is that, as crop prices rise, it becomes economically viable to invest in the conversion of additional tracts of land. Economists have used historical data to show that a correlation exists between commodity prices[83] and cultivated area in many countries. For instance, the linkage has been demonstrated in Brazil by Barr et al. (2010) and Morton et al. (2006). The effect has also been shown using the instrumental variable technique discussed in section 3.5.1 by Roberts and Schlenker (2010) and Berry and Schlenker (2011).

The strength of the connection between price and area expansion is not always the same everywhere—in particular, government policy has traditionally influenced the development of agriculture in various regions. In Europe, the Common Agricultural Policy was the key driver of how much area remained under planting for several decades, with policies such as set-aside provisions insisting that some areas should be left untilled.[84] Government in Europe also insulated agriculture from the demand side of the market, with subsidies resulting in the creation of the infamous 'grain mountains' and 'wine lakes.'[85] Policy has had a similarly dominant role in the past in the United States: Houck and Ryan (1972) assert that from 1948 to 1970 government intervention explained most of the variation in planted corn area, with only a minor role for price.

More recently, however, it is generally agreed that market effects have become increasingly important in both the EU and the United States, as agricultural policy moves away from interventionism. Taheripour, Tyner, and Wang (2011, p. 11) conclude that "in previous decades, crop acreages (distribution of cropland among alternative crops) were much more responsive to changes in government programs," but "it seems that farmers now respond to the relative crop prices more than what we observed in the past."

3.5.1. Area on price elasticity

In the economic literature, as for price-induced yield change, the strength of the area response to price is characterized as an elasticity (the 'own-price area elasticity'). However, the literature supporting any particular value for the elasticity of area expansion

83 Remember that this report talks about prices as a shorthand for expected returns. It is not necessarily true that when prices increase, returns to farmers increase well. For example, if fertilizer prices increased enough, then feedstock prices would increase but returns to farmers would fall—potentially resulting in a contraction of planted area rather than an expansion. However, discussions about ILUC are explicitly concerned with a demand-led price increase, and this price change, with all other things being equal (as is the case in the economic modeling), will necessarily result in a corresponding increase in farm net returns.

84 U.K. Department for Environment Food and Rural Affairs: http://www.defra.gov.uk/statistics/foodfarm/enviro/observatory/set-aside/. However, this subsidy has been halted as of 2007 to bring down wheat prices (see http://www.telegraph.co.uk/news/uknews/1564327/Set-aside-subsidy-halted-to-cut-grain-prices.html).

85 See, for example, http://www.nytimes.com/2009/01/22/world/europe/22iht-union.4.19606951.html

has been weak. Al-Riffai, Dimaranan, and Laborde (2010b, p. 92) pointed out that "there are no robust estimates from the econometric literature because of the complexity of the linkage and the highly fragmented data available for land use in deforested regions, the lack of a continuous time series on local prices, and more importantly, land rent, when they exist." In fact, this one study decided against using area expansion elasticity values from the literature at all and instead assigned them based on a set of assumptions.

The existing literature has also tended to be focused on the United States, leaving an even greater paucity of evidence for the rest of the world. Studies generally cover only developed nations and Brazil (where the issue of deforestation has been of particularly high interest; e.g., Barr et al., 2010; Gurgel, Reilly, and Paltsev 2007; Sohngen and Mendelsohn 2007). According to Barr et al. (2010, p. 2), GTAP, in its impact analysis for California, takes its elasticity value from a study (Ahmed, Hertel, and Lubowski 2008) that calculates only the change in cropland expansion in the United States. Basing the modeling of all regions on data applicable only to the United States constitutes a major limitation in the modeling. Similarly, the literature from which IFPRI-MIRAGE (modeling ILUC for the European Commission) takes its elasticity of area expansion is also largely based on data from developed countries (Laborde and Valin 2012, p. 11, which cites Barr et al., 2010; Roberts and Schlenker 2010; and OECD 2001).[86]

Using developed world elasticities to represent the developing world is potentially problematic, as one might reasonably expect land use dynamics to be different between developed and developing countries. On the one hand, developing countries are prone to having weaker policies to control deforestation and land conversion, and these policies may be less well enforced where they do exist. On the other hand, developed countries like the United States are well linked into global agricultural markets as exporters, and, as stated above, government policy has become less important than market signals in recent years. In a country like India, by contrast, land use may be much less connected to world prices. Barr et al. (2010) found the elasticity of area expansion to be higher in Brazil than in the United States, while Gurgel, Reilly, and Paltsev (2007) calculated higher area expansion elasticities in developed nations versus the developing world.

Roberts and Schlenker (2010) aim to fill some of the data gaps around area expansion elasticities by using an instrumental variable approach to analyze both supply and demand elasticities for the caloric sum (total digestible energy content) of the four largest food crops: wheat, corn, rice, and soy. In contrast to many commentators who have argued for a strong yield response (see section 3.2.2), they hold that the yield response, in the short run at least, is tiny compared with the area response. Therefore, they say, the supply elasticity is representative of the global area elasticity. Overall, they find that supply elasticity is likely higher than demand elasticity, by a factor on the order of between 2:1 and 3:2—that is, they would expect area increase to supply up to twice as much feedstock as demand reduction. Their supply elasticity estimates (intended as a proxy for area elasticity because they contend that yield response is negligible in comparison) are on the order of 0.1; the demand (food consumption) elasticities are on the order of 0.05 or a little higher. Berry and Schlenker (2011) provide further support for a global short-run area elasticity around 0.1, with no significant short-run yield response.

86 The area expansion elasticity used in Laborde and Valin (2012) was likely taken from the calculation for the United States from Barr et al. (2010) rather than adopted from the Brazilian value.

Berry and Schlenker provide more distinctions between countries, which is relevant to the question above of how area elasticities vary between regions. For both the United States and Brazil, which are principal agricultural exporters firmly plugged into the global market, they find much higher area elasticities than the global average: about 0.3 for the United States and about 0.4 for Brazil. For soybeans specifically, a key crop in the ILUC debate, cultivated land area in both Argentina and Brazil was very elastic with respect to price, with calculated elasticities of around 1.2. For China, on the other hand, land area was not significantly price elastic. Because (with or without the Armington assumption) the United States and Brazil are likely to be critical for ILUC evaluation, the strong elasticity of area expansion found in these countries could suggest that ILUC is being underestimated in those regions.

The evidence suggests that applying a single elasticity of area expansion to the whole world would likely be a source of error in the modeling of ILUC and would result in area expansion being predicted in the wrong regions, significantly affecting results. Generalizing from the United States will lead to an over- or underestimation of land use changes that would probably vary from country to country. For Brazil in particular, the evidence suggests that area expansion elasticity should be high, and since Brazil is a major producer of both soy and sugarcane, this could have important implications for calculating the indirect land use change implications of those feedstocks in particular.

3.5.2. Area expansion in the modeling of indirect land use change

While the econometric analysis of historical behaviors gives numerical elasticities, these numbers are not always directly input as parameters in the models. Studies on historical land extension may not give the resolution necessary for direct use as modeling parameters; for example, in the cases of Berry and Schlenker (2011) and Roberts and Schlenker (2010), it is impossible to distinguish between crop switching (the crops studied replacing other crops) and expansion onto entirely new land. More specific, regionally comprehensive estimates of elasticities of area expansion simply do not exist in the literature to the best of knowledge. In the absence of high-quality historical data, some models rely on the use of theoretically justified functional forms. GTAP, for one, determines the rate of cropland expansion using a system of 'constant elasticity of transformation' (CET) functions. These CET functions provide a framework for land to shift from one use to another (still based on elasticities, in this case elasticities of transformation from one land use to another as comparative rents change) but do not require the input of some overall land expansion elasticity, although in principle it would be possible to calibrate the CET structure to try to reproduce historical observations of area expansion elasticities. The constant elasticity of transformation (CET) relates the demand for one land type (e.g., cropland) to the demand for a second land type (e.g., pasture), with the transfer of land from one type to the other determined by changes in their relative land rents.

Within GTAP, it is possible to assign different elasticities to swaps of different land types. For example, in Golub et al. (2006, pp. 19–20), the greatest elasticity is assigned to switching between crops; this means that if the price of corn rises, its cultivation will be more likely to expand onto other cropland like soy or cotton than onto an entirely different category like pasture or forestland. (Crop switching is discussed in section 3.3.1.) A lower elasticity is assigned to the transformation of land between agriculture and forestry. The overall elasticity of area expansion for a given crop then becomes a composite of the area of expansion onto other cropland, pastureland, and managed forestland. If these values are different by region, the implied overall elasticity will depend on hundreds of model parameters.

The CET approach can only be applied to land for which one can define an associated price or rental rate (managed land); it cannot be readily applied to unmanaged land for which there is no current monetary revenue (e.g., much of the world's forests). For this reason, GTAP largely ignores unmanaged forests, which some commentators have argued is a major flaw in the model. If expansion of the agricultural frontier into unmanaged land is a likely result of increased crop prices, then it is indeed a severe limitation that GTAP is unable to model that activity.

To model expansion onto unmanaged land, one must apply some sort of additional assumption or alternative approach. For instance, one could allow the economic model to use the CET structure to determine transfers between varieties of managed land but also permit the managed land area to expand onto natural land by using historical data to calculate a price elasticity of total managed area. This would not require referencing the price or rental rate of the land onto which expansion is occurring. IFPRI-MIRAGE, modeling for the European Commission, uses this type of approach (Laborde and Valin 2012, Table 10; Al-Riffai et al. 2010a, p. 21). Like Golub et al. (2006), this version of MIRAGE applies different levels of CET nesting depending on the type of land conversion, so that crop switching between highly substitutable crops is the most elastic (has the greatest CET and the least resistance to conversion), crop switching between less substitutable crops is less elastic, and expansion into other land covers is the least elastic (Laborde and Valin 2012, Table 10)—but the total managed area also increases.

To repeat, the size of the cultivated area increase in the economic models is a function not only of the elasticity of area expansion itself but also of the elasticity of other responses, so that it can be challenging to draw conclusions about how elastic area is in a given model simply from looking at the overall land expansion results. Nevertheless, it is interesting to compare the overall expansion of cultivated area in different model scenarios to the gross requirement for additional feedstock (Figure 3.9).

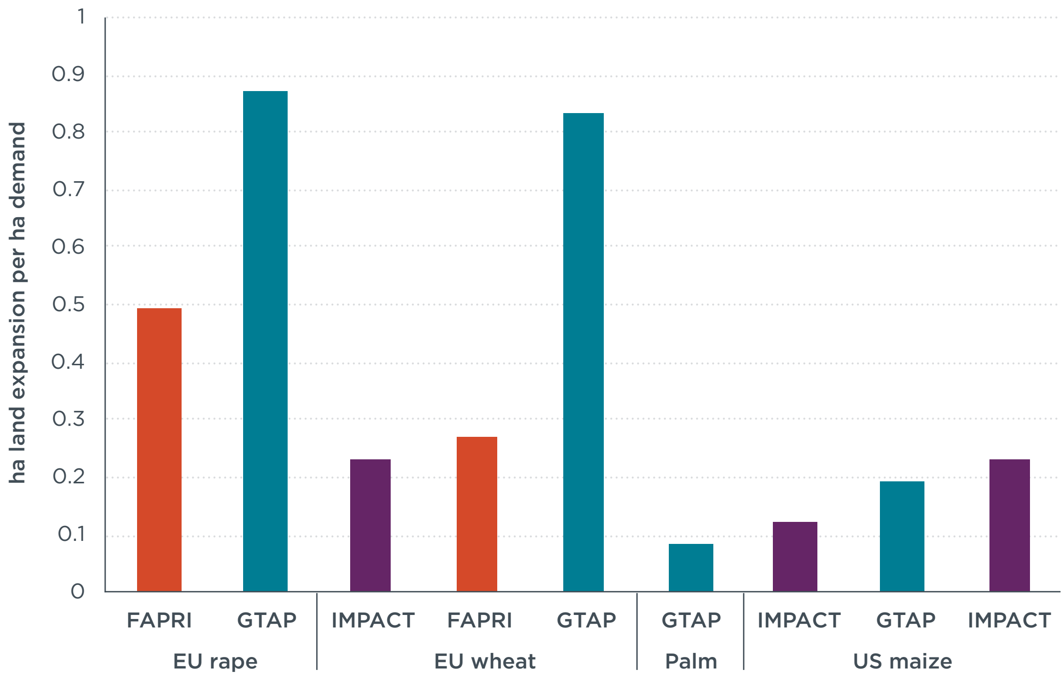

Figure 3.9. Estimates of the hectares of land use expansion required per hectare of gross biofuel land demand

Source: Edwards, Mulligan, and Marelli (2010)

The Joint Research Centre's gross feedstock area requirement is based on the average yield across all crops in each model baseline. In general, each hectare of land required to grow the biofuel to meet increased demand in the economic models compared by the JRC requires much less than one hectare of total new land conversion. For the GTAP palm oil scenario, because of the high yield of palm as well as other characteristics that reduce ILUC, less than one tenth of a hectare of land conversion is needed for every hectare of gross demand.

In general, a lower 'hectare per hectare' requirement will mean lower indirect land use change, but, as discussed in section 3.6 below, this is not always the case. Laborde (2011a, p. 71) shows that the total area requirement for a given feedstock and the carbon emissions per hectare of the land use change driven by that feedstock can be entirely independent (Figure 3.10).

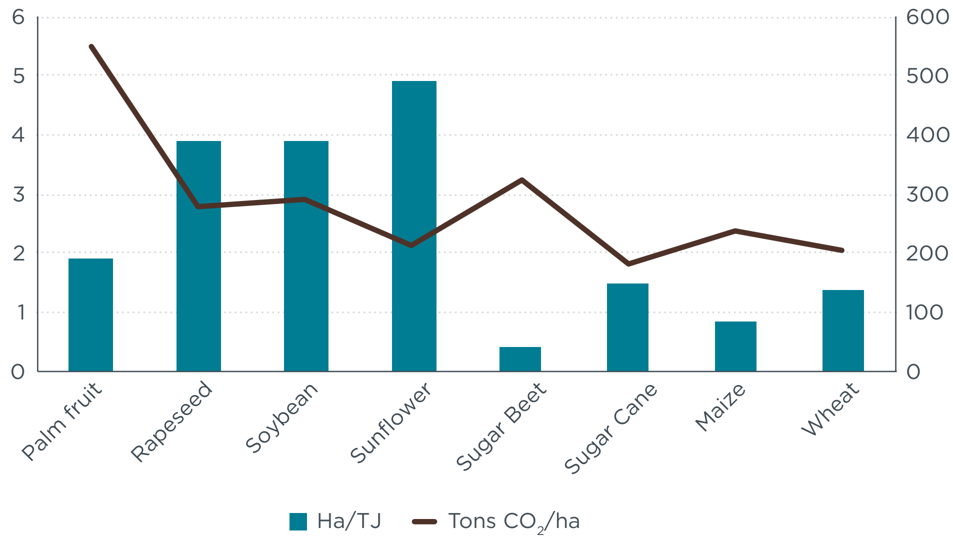

Figure 3.10. Land requirement per unit of energy versus carbon intensity of land use change for various feedstocks

Source: Laborde (2011a)

Note: Left axis and bars show the hectare requirement per terajoule of biofuel in MIRAGE modeling, while the right axis and line show the carbon emissions per hectare in each case

Notably, as with the GTAP modeling for JRC, Laborde predicts a lower overall land use change for palm oil biodiesel than other biodiesels, but because of the high carbon emissions of land converted in Indonesia and Malaysia, in the overall results of this modeling all of the biodiesels come out similarly. Malins (2011b) points out that with improved estimates of peat emissions intensity, the carbon emissions per hectare of oil palm expansion would be even higher, giving palm oil the highest ILUC factor despite its relatively small tillage footprint.

3.6. CARBON STOCK OF NEW LAND

The final factor that affects estimates of indirect land use change emissions is the magnitude of the land use change emissions factors themselves, representing the change in carbon stock when new land is converted. This refers to how much carbon is emitted into the atmosphere when natural land is converted to cropland to grow more food. Conversion of any type of land to cropland almost always releases carbon and potentially other greenhouse gases as well. While useful wood may be harvested, other tree biomass, shrubs, and grass are either burned or left to decay, and some carbon from the soil will be lost as well. Plants are about half carbon by dry weight: burning two kilograms of dry wood will release one kilogram of carbon into the atmosphere. Each carbon atom released will combine with two oxygen atoms from the air to form carbon dioxide, so one kilogram of carbon will become about three and a half kilograms of CO_2. Clearing one hectare of Amazonian tropical rain forest to create new cropland releases about 200 metric tons of carbon,[87] resulting in approximately 700 tons of CO_2 being emitted (IPCC 2006a, pp. 4.49, 4.53). That is roughly equivalent to driving an SUV for two million kilometers or running a typical coal plant for four hours—just to create one hectare of cropland, enough to supply soy biodiesel to run an

87 Including roots, dead wood, and soil carbon.

SUV for four thousand kilometers per year, one-fifth of the distance driven per year by a typical American.[88]

On the other hand, not all natural land is as carbon rich as the Amazon. Converting the Canadian boreal forest, for example, would cause the release of only about one-third as much carbon as the rain forest, or 235 metric tons of CO_2 per hectare,[89] while converting grassland would emit even less (a typical value would be around 37 tons of CO_2 per hectare).[90] Thus, which type of ecosystem is replaced by cropland makes an enormous difference to the magnitude of land use change emissions. It should be noted that no land is carbon-free. Even abandoned agricultural land in Europe, which is sometimes discussed as if it were a carbon-free source of land for biofuel production, would still typically cause the immediate release (ignoring the question of forgone sequestration; see section 3.6.3) of around 270 tons of CO_2 per hectare [91] into the atmosphere because forests have already regrown (or are in the process of regrowing) on much of these lands. Additionally, these abandoned fields tend to have considerable biodiversity, more than one might expect, which would also be lost if they were cultivated once more (Prévosto et al. 2011). Recultivating abandoned land in the United States, especially on Midwestern prairies, would have lower carbon consequences but still would probably result in the emission of at least 60 tons of CO_2 per hectare upon conversion.

Overall, the following three questions determine how much CO_2 is emitted from land conversion:

1. What type of ecosystem is destroyed?
2. How much carbon was there?
3. How much of that carbon is emitted into the atmosphere?

Once an economic model has determined how much land must be converted to meet demands for fuel, food, and fodder, the accuracy and precision with which it answers these questions drive the final result. For instance, if a model underestimates total carbon losses from land conversion by half, then its calculated ILUC factor will be 50 percent too low, which is more than enough to turn a biofuel that delivers no net carbon savings into one that appears to be sustainable.

3.6.1. What type of ecosystem is converted?

The first question is what type of ecosystem we expect to be converted to make way for additional cropland. Having defined categories of land (such as cropland, pasture, grassland, shrubland and forest), a model must consider where cropland expansion will occur among these categories. Expectations about which ecosystems will be converted and the carbon consequences of that conversion will vary from region to region and may also vary from crop to crop. For instance, if land use expansion occurs in Brazil, will it happen on grassland or forestland? Is there reason to think

---

88 U.S. Department of Transportation, http://www.fhwa.dot.gov/ohim/onh00/bar8.htm

89 Including roots, dead wood, and 25 percent soil carbon release. Carbon stocks from the IPCC (2006a, p. 4.54; 2006b, p. 2.13), assuming spodic soils according to http://www.radford.edu/~swoodwar/CLASSES/GEOG235/biomes/taiga/taiga.html

90 Includes 25 percent soil carbon release and shows median global values for grassland vegetation and soil carbon stocks. Vegetation carbon stocks from the IPCC (2006b, p. 6.27; 2006c, p. 2.31) and soil carbon stocks from the Harmonized World Soil Database.

91 See Appendix B for calculations.

that expansion of sugarcane, largely grown further south, away from the remaining forested areas, is less likely to result in forest conversion than soy expansion is? Given the differing carbon stocks between these categories, the land extension coefficients (LECs), the fraction of cropland expansion that occurs on each type of land, can have a large impact on the results.

If future patterns of land use change echo past land use change, then historical analysis can help inform expectations. For example, if 25 percent of cropland expansion in Brazil replaced forest over the past decade and 75 percent occurred on grassland, then one might assume that the same percentages would apply for cropland expansion over the next year. One such analysis was undertaken by Winrock International, which produced a global dataset of LECs based on changes in land cover from 2001 to 2007, based on analyzing the U.S. National Aeronautics and Space Administration's Moderate Resolution Imaging Spectroradiometers (MODIS) satellite data. This dataset was used by the EPA for its RFS2 Regulatory Impact Analysis (U.S. EPA 2010a, p. 394) and also in IFPRI's modeling with MIRAGE (Al-Riffai, Dimaranan, and Laborde 2010b, p. 22).[92] The Winrock-MODIS LECs provide a comprehensive set of data for ILUC modeling, but they have also been widely criticized over questions of accuracy owing to the occurrence of false positives (Marelli, Mulligan, and Edwards 2011, p. 18).

The problem with this satellite-based approach is that, while the land classifications in each year may individually have acceptable accuracy, when they are compared to look for changes, any errors can be compounded. So if, for instance, 5 percent of land classification in each year for a given region were erroneous, there would still be 95 percent accuracy, and the general understanding of land uses would be reasonably good. However, in comparing for changes between two different years, up to one in ten grid cells would be identified as changing land use when no change had actually occurred. If real land use changes only occurred on 5 percent of the land, there would be a strong risk of a situation in which false positives outnumbered actual shifts. In that case, the LECs would be determined by the most common classification errors rather than by genuine land use change. The global land use identification accuracy reported for the MODIS mapping used by the EPA is 72 percent, and it has been noted by Holly Gibbs (Marelli, Mulligan, and Edwards 2011) that, according to MODIS differentiating by Winrock, 94 percent of all shrubland changed land cover type over the seven-year period assessed. That seems implausibly high and compares to an estimate of 9 percent by Gibbs using Landsat change detection. Nevertheless, despite the potential challenges of satellite data such as MODIS, the basic principle of using historical trends is a reasonable approach. For example, for palm plantations on peatland, Miettinen et al. (2012) use high-resolution satellite imaging to demonstrate that the rate of expansion has been accelerating between 1990 and 2010—the combination of high identification accuracy and significant actual land use change makes these results more robust.

An alternative (or sometimes complementary) system that is used in economic modeling to determine types of land conversion is to predict land use changes based on the value of land, with reference to existing land uses in the same region. If cropland expansion is anticipated in a certain region, and the model indicates both

92 IFPRI's final report to the European Commission (*Assessing the Land Use Change Consequences of European Biofuel Policies*) generally did not provide detail on how land use change emissions were calculated. This chapter assumes that the methodology for the European Commission report was consistent with other recent IFPRI publications using the MIRAGE model.

that it is cheaper to purchase or rent shrubland than forestland and that significant areas of shrubland are available, then there is a good chance that farmers will expand onto shrubland. The details of the decision making in models that carry out these types of assessment, like MIRAGE and GTAP, are determined by constant elasticity of transformation equations, which also include elasticity values that gauge the ease with which land moves between different types. As an example, GTAP assumes that different types of cropland are more readily interchangeable than cropland and pasture, or than cropland and forest, given the comparable rents and availability. This approach only works for managed lands to which a price can be attached; unmanaged lands, which do not have a well-defined land rent, cannot be modeled this way. IFPRI's MIRAGE therefore combines the use of CET functions for managed land and Winrock's MODIS coefficients for expansion into unmanaged areas.

A third approach goes beyond the simple economics-based determination and uses a broader set of land characteristics to forecast which land areas will be converted. The Joint Research Centre (Hiederer et al. 2010) employs one type of spatial allocation methodology; another has been developed by Winrock for predicting farmed land expansion in Southeast Asia for the EPA's analysis of land use change from consumption of palm oil biodiesel. These methodologies take into account characteristics such as rainfall, slope, soil quality, proximity to roads, proximity to existing production areas, and so on to predict which areas will be targeted when conversion occurs. If data are available, the allocation methodologies can be calibrated by comparing their predictions with observed patterns of prior land use change. These approaches therefore assume not that the categories of land converted will necessarily be the same as in the past but rather that the decision-making process for farmers expanding cultivated area now will be based on the same considerations as in the past.

LECs, aggregated into forest or grassland where models have more detailed categorizations, are shown for the various models in Figure 3.11. These LECs are taken from different scenarios and in some cases reconstructed from published documentation and so should not be directly compared. Still, it can be seen that they all predict a greater proportion of land use change occurring on grassland than in forest.

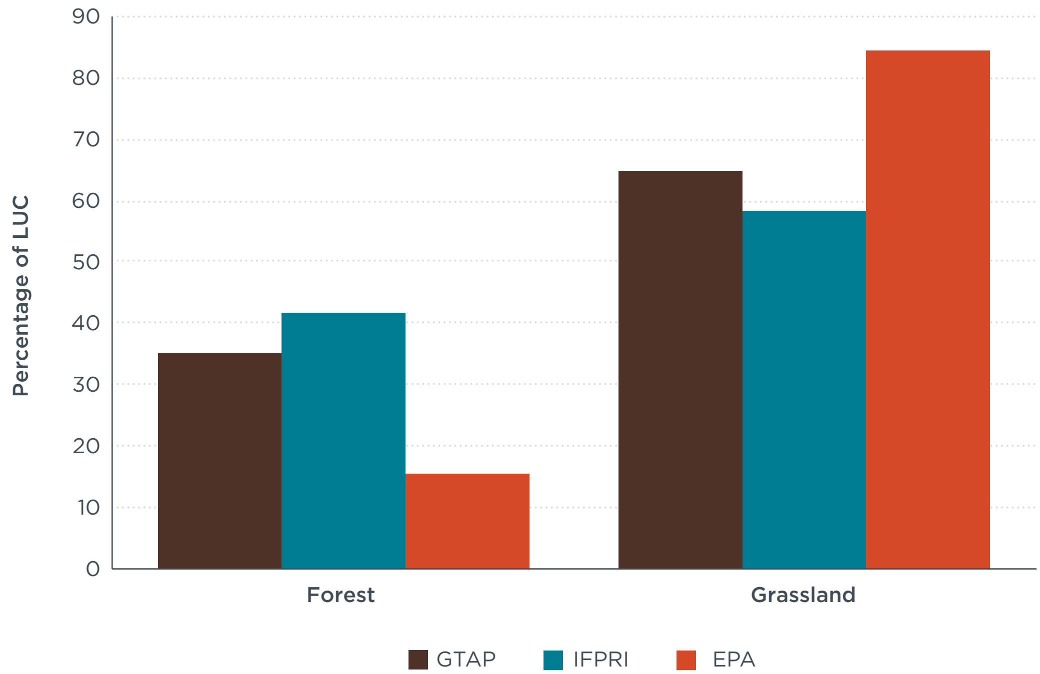

Figure 3.11. World average land extension coefficients (the percentage of land use change occurring in each ecosystem type) for various models

Note: The percentage of cropland expansion occurring in forest in GTAP was provided in Tyner et al. (2010, p. 46, using 2006 as a baseline); it is assumed that the remaining expansion occurred on grassland (including pasture). The proportion of cropland expansion occurring in forest versus grassland was given in Laborde (2011a, p. 48). The EPA did not provide world average land extension coefficients in its documentation; these were crudely reconstructed based on LECs given for various countries in U.S. EPA (2010a, p. 392), as well as a rough description of which countries witnessed crop expansion (Harris, Grimland, and Brown 2008, p. 13).

Forests store more carbon than shrublands, which store more than grasslands, but economic models do not always account for all these different land types. GTAP, as used for the ILUC factors for California's Low Carbon Fuel Standard, includes only grassland and managed forest (Tyner et al. 2010, p. 34; Hertel et al. 2010b, p. 27). In contrast, FASOM, which the EPA used to model land use changes within the United States, is highly detailed, covering 25 different forest species types and 18 different forest management intensities (Adams et al. 2005, p. 41). Most models predicting land use change as a result of biofuels demand have a limited number of categories. This is understandable, given the lack of data and modeling limitations, but it can still inject a substantial margin of error into the results.

To understand why this matters, take Brazil as an example. Ninety-five percent of Brazil's forests are tropical/subtropical, moist broadleaf forests, which have around 112 metric tons of carbon per hectare (tC/ha) in biomass, while 5 percent of Brazil's forests are dry broadleaf forests, which have around 83 tC/ha (WWF 2012; IPCC 2006a, pp. 4.49, 4.63). Thus, the average amount of carbon in Brazil's forests is (0.95 x 112) + (0.05 x 83) or 111 tC/ha. If land conversion were to occur uniformly around Brazil's forests, this is how much carbon would be emitted, on average, and this is the value that would be assigned to the 'forest' category in GTAP. But say, hypothetically, that it is easier for farmers to convert Brazilian dry forest to cropland than moist forest, with 80 percent of land conversion occurring in dry forest. Now the average amount of biomass carbon released from land conversion would be (0.2 x 112)

+ (0.8 x 83) or 89 tC/ha. In this example, failure to distinguish the two different forest types would have resulted in a 25 percent overestimation of carbon emissions.

3.6.2. How much carbon was there?

It is understood that forests store more carbon than grasslands, but how much more? And how do these carbon stocks change with ecosystem type?

As mentioned at the beginning of this section, there is a range of carbon stocks even within forests. According to the IPCC, tropical rain forests can hold as much as 145 tC/ha in biomass. Drier forests typically store less carbon (60–100 tC/ha for tropical dry forests; IPCC 2006a, p. 4.53), while forests in cooler climates also tend to hold lower carbon stocks (temperate continental forests store about 60 tC/ha, and boreal forests only have 35 tC/ha; IPCC 2006a, p. 4.54). Grasslands have much less carbon in biomass than forests, with only about 2–4 tC/ha (IPCC 2006c, p. 6.27). There can be large variations even within single types of forest in relatively small regions. For Malaysian primary forest in one region of the state of Sarawak, Proctor et al. (1983) identify a range of 105–325 tC/ha. If there is some unknown systematic bias for agriculture to expand onto either the most or least carbon-rich plots of land, then the use of regional averages could introduce a substantial skew into emissions accounting.

There are also many categories of ecosystems that are difficult to classify: shrublands, savannahs, woodlands, etc. These ecosystems typically have some small trees but not enough to be called a forest. For those models, such as GTAP or MIRAGE, that lack a 'shrubland' category, it is difficult or impossible to capture accurately how much carbon is emitted if these types of areas are significant sources of land for conversion. Carbon stocks for the major models are shown in Figure 3.12.

Although the land extension coefficients for most of the major models do not differentiate between forest types or between grassland types, it is still possible to use carbon stocks specific to a location. Both GTAP and MIRAGE also vary assumptions about the carbon stock in each category of land based on the region in which conversion occurs.

As well as being stored in biomass, carbon is also concentrated in the soil. Over decades to millennia, rotting biomass is captured in soil, which stores a hefty amount of organic carbon. Soils can store anywhere from 10 to 130 metric tons of carbon per hectare according to the IPCC (2006b),[93] which can be as much as or more than the carbon stored in biomass in any particular ecosystem. There is not a clear difference in carbon storage in tropical soils versus those in cooler climates, but grassland and pasture soils typically sequester more carbon than forest soils (Guo and Gifford 2002; Don, Schumacher, and Freibauer 2011). Thus, where cropland expansion occurs affects how much carbon is emitted from the soil as well. This also implies that the difference in carbon loss between converting forest and grassland is less than one might expect from looking at the biomass figures.

Some carbon in forests is also stored in dead wood and litter. This fraction likely amounts to about 10 percent of total biomass in tropical forests (Delaney et al. 1998) but may constitute up to 40 percent of total biomass in temperate forests (Litton et al. 2004; Turner et al. 1995). Nearly all of the major models ignore dead wood and litter in their estimates of carbon stored in biomass, leading to an underestimation of land use change emissions across the board.

93 For organic peat soils, the number could be much greater.

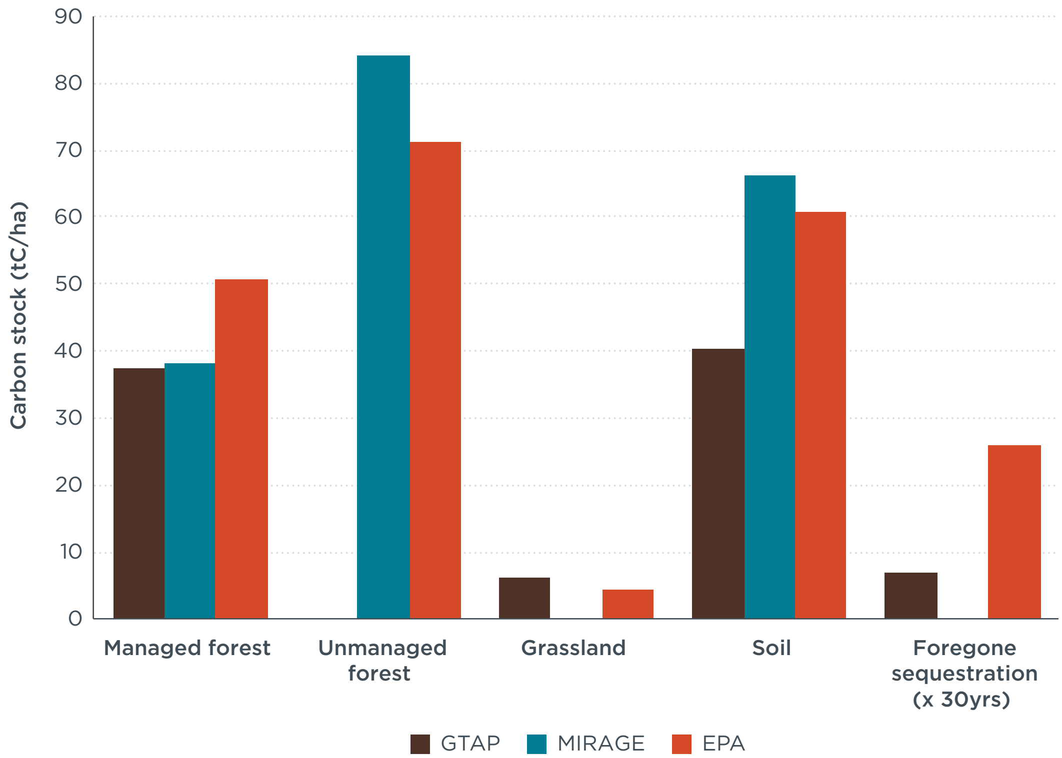

Figure 3.12. World average carbon stocks for managed and unmanaged forest, grassland, and soil, and carbon lost from forgone sequestration

Note: Amortized over 30 years. Unmanaged forest does not exist in GTAP. Classification of unmanaged versus managed forest may differ between the models. It was not possible to determine whether MIRAGE includes grassland and forgone sequestration. All carbon stocks shown here were reconstructed based on the description of the methodology in each model's documentation.

3.6.3. How much carbon is removed?

It is clear that most biomass carbon must be emitted upon land conversion. Certainly, all grassland biomass is lost. On the other hand, some forest biomass persists in the form of wood products. Typically, when forests are cleared, the most valuable wood is harvested and sold for profit. Carbon in a wooden table, for instance, remains sequestered for as long as that table exists. One study estimates that 10 percent of cleared biomass remains stored in harvested wood products or landfills[94] for the developed world after 30 years but at most only 3 percent in the developing world (Searle and Malins 2011).[95]

Between 20 percent and 40 percent of soil carbon is lost from forests converted to cropland,[96] up to 52 tC/ha.[97] Grasslands and pastures may lose even more carbon upon conversion to cropland, up to 59 percent (Guo and Gifford 2002). An estimate of average soil carbon loss with cropland expansion over all ecosystem types is around 30

94 Over a period of 30 years.

95 In a different approach, Earles, Yeh, and Skog (2012) found that about 40 percent of tree biomass remains sequestered in Western countries after 30 years and less than 5 percent in developing countries; these estimates are likely higher because this study did not examine biomass loss in the understory, dead wood, or litter.

96 Don, Schumacher, and Freibauer (2011); Luo, Wang, and Sun (2010); Takahashi et al. (2010); Guo and Gifford (2002); Murty et al. (2002); Davidson and Ackerman (1993).

97 Using carbon stocks in IPCC (2006b, p. 2.31).

percent.[98] Most of the major models assume soil carbon losses not very different from this; for example, GTAP has assumed that 25 percent of carbon is lost from all soils with cropland expansion (Tyner et al. 2010, p. 34). Organic peat soils are a special case, with ongoing emissions long after conversion (see the following section).

There is one more source of 'carbon emissions' involved in land use change. Globally, plants are sequestering more CO_2 than they emit; this gross carbon sink amounts to approximately one-third of total anthropogenic CO_2 emissions.[99] Thus, when cropland that might have been abandoned under a baseline scenario must remain in cultivation because of biofuel demand, the carbon that would have been sequestered by the regrowing forest must be considered 'emitted.' Similarly, when a growing forest is converted to cropland, not only is the existing carbon released but also the CO_2 that would have been sequestered over time had the forest remained intact; this is termed 'lost sequestration.' These estimates are shown for the various models in Figure 3.12. In the earlier example of cultivating abandoned cropland in Europe, the emissions from converting this type of land would rise from around 270 tCO_2/ha to around 510 tCO_2/ha when forgone sequestration is included and summed up over a 30-year period. Forgone sequestration can clearly be important to take into account in some cases.

It would not be accurate to say that there is always a net carbon emission following land conversion. There are certain special cases where biofuel feedstocks could store more carbon than the biomass (and soil) that existed someplace previously. For example, significantly more carbon would be stored in an oil palm plantation replacing an *Imperata*[100] grassland in Indonesia (Casson, Tacconi, and Deddy 2007); in this case, the emission factor from land conversion would actually be negative. It is also possible that replacing natural grasslands in some regions with high-yielding perennial grasses for biofuel production (e.g., 'energy cane') could result in a net increase in carbon storage. On the other hand, there could be many cases in which perennial biofuel feedstocks would be low yielding on poor soil, and, after accounting for the period of low carbon storage around harvest and regrowth, there would be a net carbon loss from conversion.

3.6.4. Peat soils

There is one special type of soil from which carbon emissions following conversion are higher than for any other category of land: peat soils. Peat is dead organic matter, preserved in oxygen-low water and accumulated over centuries; because it is concentrated biomass, it is very high in carbon (Murdiyarso, Hergoualc'h, and Verchot 2010; Page et al. 2011b). As farmers transform peatlands into oil palm plantations, they drain the water, exposing the peat to air and allowing centuries' worth of peat to rot, releasing vast amounts of CO_2. An ICCT report (Page et al. 2011a) estimates that peat drainage releases 29 tC/ha per year (or 105 tons of CO_2 equivalent per hectare per year) when averaged over 20 years, or 573 tC/ha over 20 years combined. This is about two and a half times the amount of carbon that is released from cutting down the average tropical rain forest (IPCC 2007, pp. 2.31, 4.49, 4.63) and more than the 305 tC/ha reported for the maximum rain forest carbon stock in the same region by Proctor et al. (1983). Also, peat emissions are ongoing rather than taking place immediately

98 Calculated based on a review of Don, Schumacher, and Freibauer (2011); Luo, Wang, and Sun (2010); Takahashi et al. (2010); Guo and Gifford (2002); Murty et al. (2002); Davidson and Ackerman (1993).

99 Pan et al. (2011); Reich (2011); Malhi (2010); Myneni et al. (2001).

100 An invasive grass in Indonesia.

after conversion. For a fairly typical three-meter-deep bog, emissions could continue for another 100 years afterward (Fargione et al. 2008).

Peat soils can be particularly important in considering ILUC emissions from biodiesel production because peat degradation is strongly connected to palm oil expansion. A substantial proportion of oil palm in Indonesia and Malaysia is planted in peat. Indeed, the ICCT (Miettinen et al. 2012) demonstrates that the rate of conversion of peat to oil palm plantations has been accelerating steadily since 1990, while the Joint Research Centre (Edwards, Mulligan, and Marelli 2010) concurs with the ICCT that at least one-third of palm expansion in the next decade can be expected to occur on peat.

Most models, however, significantly underestimate peat emissions from oil palm expansion. For instance, IFPRI's 2011 analysis for the European Commission assumed peat emissions to be 55 metric tons of CO_2 equivalent per hectare per year (Laborde 2011a, p. 50). The 2010 IFPRI analysis for the European Commission (Al-Riffai, Dimaranan, and Laborde 2010a) used a value of only about 19 tons of CO_2 equivalent per hectare per year. Most other models have ignored peat entirely, although recent modeling by the EPA of ILUC from palm oil includes peat emissions in line with the ICCT's findings (Page et al. 2011a). Laborde (2011a, p. 51) modeled peat emissions to account for one-third of all land use change emissions that would result from increased biofuel production. Had Laborde adopted what the ICCT considers the best estimate,[101] total land use change emissions would have increased by about one-third, eliminating the modeled carbon benefit of European biofuels policy (Malins 2011a). The magnitude of this difference highlights the importance of peatlands in the land use change debate.

3.6.5. Emissions factors in the modeling of indirect land use change

The choice of emissions factors for different land types makes an enormous difference to the ILUC factors calculated by a given modeling exercise. Assuming a few metric tons more carbon loss from the conversion of pasture could be the difference between making a biofuel look like a viable climate mitigation option and making it look worse than fossil fuels.

The Joint Research Centre (Edwards, Mulligan, and Marelli 2010) gives an example of the importance of differentiating between land types. The GTAP (using emissions factors from the Woods Hole Oceanographic Institution database) and FAPRI (using GreenAg-Sim for emissions factors) scenarios, compare the emissions derived from assuming a flat average of 40 tC/ha for all land (plus peat oxidation emissions) against the emissions calculated by the models differentiating carbon stock by land type (Figure 3.13).

101 95 tC/ha per year spread over 30 years (Page et al. 2011a).

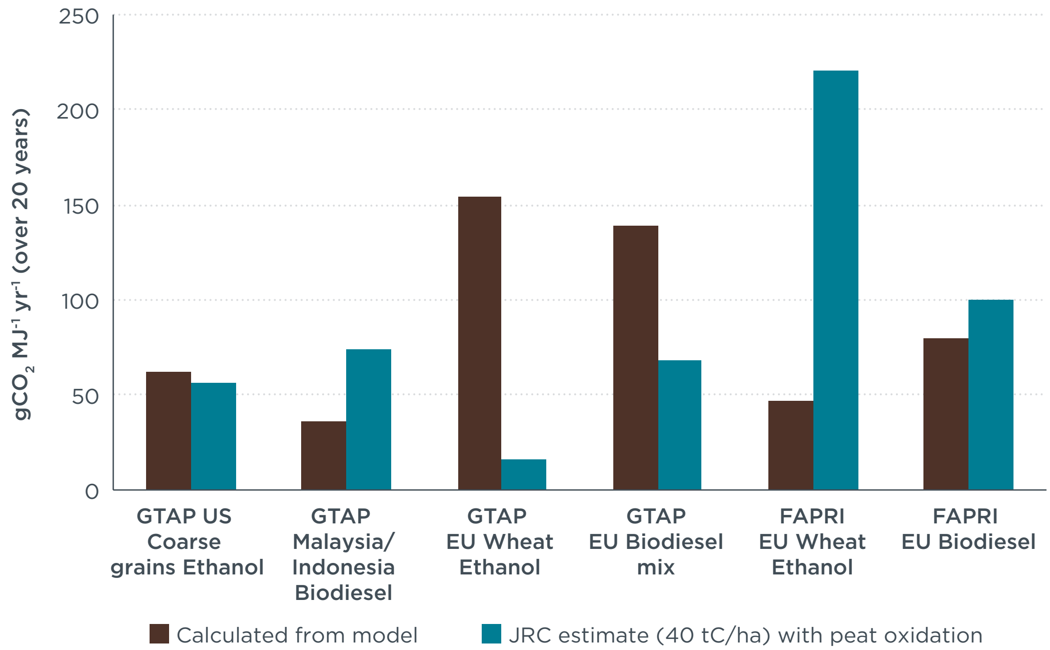

Figure 3.13. Comparing ILUC factors derived by assuming an average emissions factor for all land against differentiating by land type in the modelers own emissions calculations

Source: Edwards, Mulligan, and Marelli (2010, p100, Figure 23)

For the GTAP scenarios, the flat average emission assumption is not that far from the results predicted by the modelers—the biggest difference is for Malaysian/Indonesian palm oil, where (slightly surprisingly) the use of Woods Hole Oceanographic Institution emissions factors by land type actually gives a lower ILUC factor by about 33 percent than the Joint Research Centre flat estimate. Part or all of this discrepancy must be because GTAP ignores peat emissions, which have been added to the JRC estimate based on an assumption by JRC that one-third of palm expansion occurs at the expense of peatlands. Also, the inability of GTAP to model unmanaged land effectively is likely to be particularly problematic for a region like Southeast Asia, where historically much palm expansion has occurred at the expense of unmanaged forestland.

For FAPRI, the discrepancy is much larger, with the JRC number for wheat ethanol being nearly four times the number calculated with GreenAgSim, while the JRC value for rapeseed biodiesel is only half that reported using GreenAgSim. It is unclear exactly why there is such a difference between these outcomes. Presumably, such expansion as occurs in Europe would have a similar carbon footprint per hectare regardless of crop in GreenAgSim. It would follow that the model must be predicting that, in the biodiesel case, deforestation is taking place overseas (perhaps by expanding palm oil demand to replace vegetable oil in the EU food market and assuming that palm oil is a strong driver of deforestation). These differences show that the emissions factor assumptions are central to obtaining credible results.

The Laborde (2011a) and Al-Riffai, Dimaranan, and Laborde (2010) MIRAGE modeling for the European Commission gives useful examples of how important even a single emissions factor can be. Specifically, Al-Riffai and colleagues use an emissions factor for peat conversion of 19 metric tons of CO_2 equivalent per hectare per year, whereas Laborde increases this to 55 tons. Even the latter is likely to be an underestimate, and Laborde also provides a higher value of ILUC for palm biodiesel if the emissions factor of 105 tons of CO_2 equivalent per hectare per year recommended by Page et al. (2011a) is used.[102] Figure 3.14 shows the difference in the ILUC factor for palm using each of these three emissions factors in the Laborde modeling.

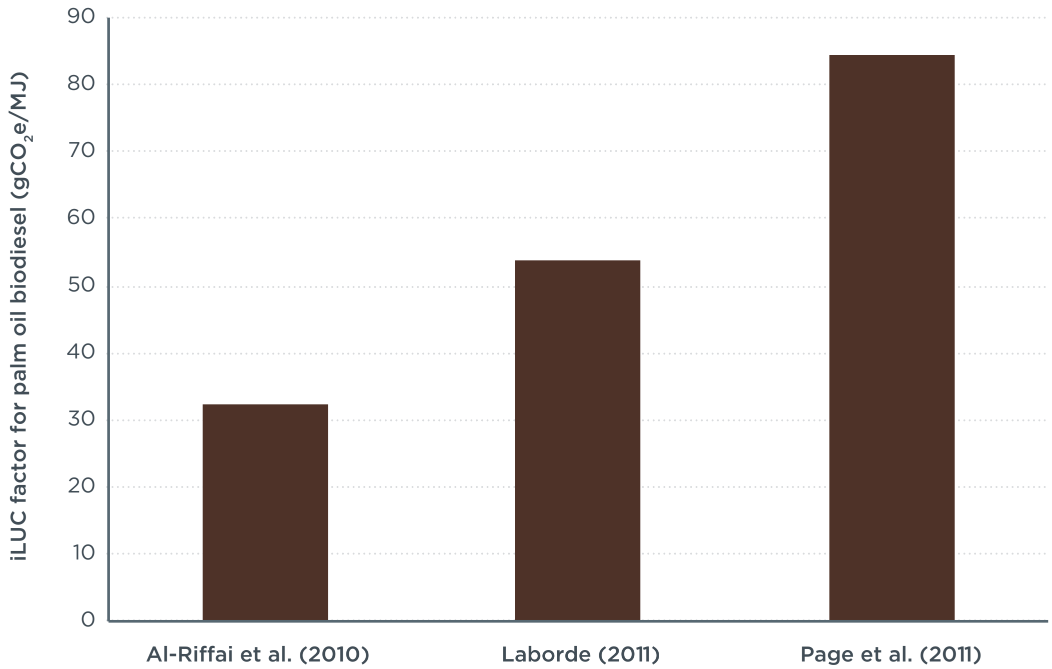

Figure 3.14. Effect of using different peat decomposition emissions factors on the ILUC factors for palm biodiesel in MIRAGE modeling

For the palm biodiesel scenario, the peat decomposition emissions are so determinative that changing that single parameter from 19 to 106 metric tons of CO_2 equivalent per hectare per year results in an increase in ILUC emissions of 260 percent.

102 Laborde (2011a), p. 53, footnote 21.

4. ILUC ILLUSTRATIONS

4.1. BIOFUEL VISIONS: ECONOMIC MODEL VERSUS MODEL OF BEST PRACTICE

One way to explore the importance of the six factors in determining the amount of indirect land use change is to use them as a frame to compare two very different visions of the land use impacts of biofuel policy. On the one hand, this chapter will consider the original ILUC modeling exercise by Searchinger et al. (2008).[103] In this study using partial equilibrium economic modeling, demand causes price to rise, and the agricultural system responds in a way that eliminates any climate benefits from biofuel use. On the other hand, there is a description in Dale et al. (2010) of "Biofuels Done Right," in which, rather than modeling ILUC, the authors outline a vision of how to structure biofuel production so that no ILUC emissions are necessary. Searchinger and colleagues conclude that ILUC renders biofuel policies ineffective, Dale and co-authors that ILUC can be avoided entirely, but the reasons they reach these conclusions can still be expressed in terms of the six factors. The comparison is detailed in Table 4.1.

Table 4.1. Comparing the scenarios mapped out by Searchinger and Dale

Factor	Assumption by Searchinger	ASSUMPTION BY DALE
Elasticity of food demand to price	Food demand is elastic to price but not as much as supply is.	There is no need to eat less, as the demand can better be accommodated in other ways.
Elasticity of yield to price	Positive and negative yield effects cancel out.	Biofuel demand drives innovation in energy crop agronomy.
Crop choices	Crop choice responds to price, and farmers are somewhat resistant to change.	Farmers choose to grow high-productivity energy crops.
Utilization of co-products	Distillers grains are returned to the feed market, reducing net corn demand by about one-third.	Co-products from energy crop production are used to feed livestock.
Elasticity of area to price	Land use globally responds to price pressures.	Area in the United States increases to meet demand, so no expansion is necessary elsewhere.
Carbon stock of new land	This reflects historical patterns of land use change, including expansion into high-carbon areas.	Only low-carbon-stock land is brought into production—there is no expansion into high-carbon areas.
Conclusion:	ILUC wipes out the carbon savings of corn ethanol (108 gCO_2e/MJ).	The system can provide both food and fuel without ILUC emissions.

In these two visions of the biofuel economy, different expectations of how the six factors come into play make the difference between U.S. biofuels policy causing substantial net GHG emissions or enabling a massive expansion of bioenergy use with no ILUC at all.

103 The 2008 paper by Searchinger et al. has been extensively discussed, both critiqued and defended, since its publication. For example, responses including a critical letter by Michael Wang and Zia Haq and the subsequent riposte by Timothy Searchinger are available from the Science website: http://www.sciencemag.org/content/319/5867/1238.abstract

Part of the distinction is that the economic study assumes that all demand signals are transmitted through price and that decision-making is only as sustainability-conscious as it has been in the past (e.g., deforestation continues at the historical rate). In the Dale conception, on the other hand, some of the outcomes cannot be achieved by price signals alone—such as the specific targeting of biofuel production on low-carbon-stock land. Most (though not all) estimates of ILUC from corn since the Searchinger study have found lower values, and most analysts would agree that the policy choices and agricultural responses outlined by Dale are an idealized case (though there are some estimates whereby ILUC is even negative). Other models tend to come out quantitatively somewhere between the two, but in each case understanding the assumptions about the parameters they use would enhance the understanding of the quantitative results.

4.2. ICCT SIMPLE MACROMODEL OF ILUC

In the real world, the six drivers of the magnitude of ILUC emissions consist of a myriad of individual microeconomic decisions about agricultural practices, what to eat, livestock diets, whether to clear land, and so forth, but at the macroscopic level one can think of them in terms of a few simple parameters. The overall price elasticity of yield, for instance, can be characterized by the response of yield to price for existing production and the yield achieved on land newly brought into production.

What follows is a simple model of the workings of each contributor to determining overall land demand, a spreadsheet model characterized by macroscopic effects: the yield effects just mentioned, the overall elasticity of supply and of demand, the average carbon content of new land, etc. This model, called the "Not So Reduced Form ILUC Model" (NoSoRFIM) in reference to the Reduced Form ILUC Model of Plevin et al. (2010), is documented in Appendix C. It goes beyond the Plevin model by explicitly considering the influences on the overall land requirement, while the Plevin model has a single parameter for land demand. Using this model, one can map out the way in which the six determinants of ILUC emissions might interact in a given scenario. Model results will be presented for the case of corn ethanol. In that case, the central parameter values used in the model have been based on an assessment of the literature and calibrated to a corn ethanol pathway with an ILUC emissions intensity estimate of 30 grams of carbon dioxide equivalent per megajoule (gCO_2e/MJ) when land use change emissions are spread over 30 years, matching the corn ethanol result in the California Air Resources Board's Low Carbon Fuel Standard (LCFS) analysis. The following subsections provide a quantitative overview of the way that each factor affects the ILUC emissions likely to result from biofuel expansion.[104] Bear in mind that some determinants that are important for corn may be less so for other feedstocks and vice versa. For instance, co-products would be much less important to palm oil biodiesel, while soil carbon accumulation might be a key question for perennial crops, and changes in food consumption might be less important when considering nonfood feedstocks.

A model like this is useful for exploring the dynamics of ILUC. However, it is not of comparable complexity to a full economic model, and the values and ranges given should be treated only as illustrative—these results could not replace the outcomes of more sophisticated modeling. The reductions in expected ILUC emissions as each factor is considered are shown in Figure 4.1.

104 Reflecting the illustrative purpose of this model, all interim ILUC factors are rounded to the nearest 5 gCO_2e/MJ, and percentage ILUC reductions are rounded to the nearest 5 percent.

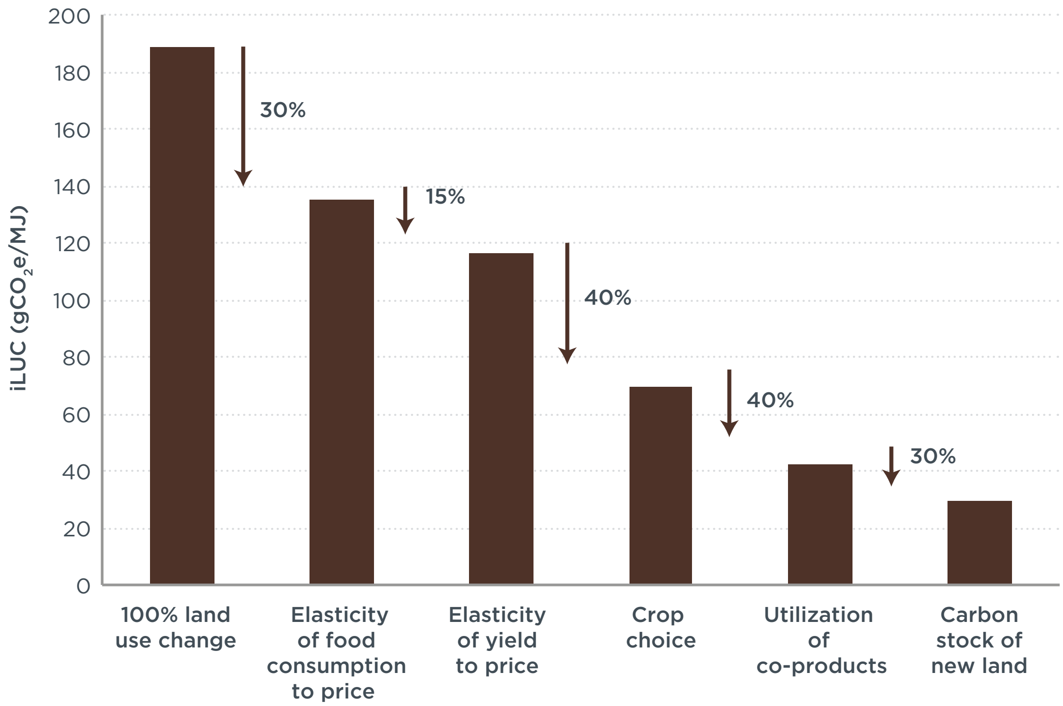

Figure 4.1. Scenario for importance of factors in determining ILUC emissions related to U.S. corn ethanol

4.2.1. Starting point: Responses other than area not yet considered, 100 percent land use change

As a starting point for the illustration, consider what the ILUC emissions intensity associated with corn ethanol production would be if all of the biofuel feedstock came from converting new land, at average yields and carbon stocks. Setting the baseline this way (a sort of maximum, or at least near-maximum, case for the magnitude of land use change) allows quantification of the other determinants in terms of the ILUC emissions intensity avoided relative to the baseline. Of course, it would be possible to choose an alternative scenario as the baseline and tell the story in a different way, for instance, starting the model from an assumption that all the necessary feedstock was subtracted from food markets. Still, once all the determinants have been 'activated,' the result would come out the same.

At a global average yield of 4 metric tons of corn per hectare, and assuming that on average 95 tons of carbon are lost per hectare following land conversion, the outcome would be an ILUC factor of 190 gCO_2e/MJ, so high that there could never be carbon savings compared with fossil fuel. This is what would happen if the only response to increased demand for biofuels were to expand area—that is, if area elasticity to price were positive but both food demand elasticity to price and yield elasticity to price were zero.

4.2.2. First factor: Food consumption falls, reducing ILUC by 30 percent

When prices increase, there is normally some response on the supply side (increasing agricultural production) but also a response on the demand side (people eat less, drink less, buy less new cotton clothing, etc.). It is assumed that the elasticity of demand is a bit lower than the elasticity of supply; still, the cutback in consumption reduces ILUC emissions intensity by 30 percent, to 135 gCO_2e/MJ.

4.2.3. Second factor: Yields change, reducing ILUC by 15 percent

On the supply side, some feedstock is supplied by greater yields as farmers respond to higher prices by increasing productivity and some by increased area. Here, yield gives one-third of the supply response and area increase produces two-thirds. On the other side of the coin, the yields on new areas of land brought into production will typically be lower than existing average yields; it is assumed that they are 15 percent lower. The overall effect of yield changes is to reduce ILUC emissions intensity by 15 percent, to 115 gCO_2e/MJ.

4.2.4. Third factor: Crop choices change, reducing ILUC by 40 percent

The ILUC calculated for 100 percent area change is based on a world average grain yield of 4 metric tons per hectare, but U.S. corn yields are much higher than this, 9 tons per hectare. It is assumed that a disproportionate fraction of the agricultural expansion will happen in the United States, so that the average yield for land where expansion actually occurs is 7 tons per hectare. Crop choices are also allowed to change throughout the agricultural system as prices adjust, with the assumption that this reduces overall demand for new land by 20 percent. Overall, ILUC emissions intensity is reduced to 70 gCO_2e/MJ.

4.2.5. Fourth factor: Co-products reduce ILUC by 40 percent

The initial calculation ignores co-products. Nearly 40 percent of the edible content of corn grain is returned to animal feed markets as distillers grains. In the United States, distillers grains primarily displace feed corn on a one-to-one basis, so this effectively reduces feedstock demand and hence ILUC emissions intensity by 40 percent, to 40 gCO_2e/MJ. Here the operating assumption is that a metric ton of dried distillers grains (DDGS) replaces a ton of corn feed and that, because the DDGS largely go back into the corn market, there is none of the bonus that would be seen if a feed product with a higher land use impact than corn was being displaced.

4.2.6. Fifth factor: Elasticity of area to price

In this illustration, the elasticity of area to price is already 'turned on' when the emissions were calculated assuming that all new production came from new land, so it cannot be switched on sequentially like the other factors. The important question is how area elasticity compares to the food consumption and yield elasticities. Area elasticity is therefore missing from Figure 4.1, but Figure 4.2 illustrates how reducing or increasing the area elasticity would affect the ILUC emissions.

4.2.7. Sixth factor: Land expansion tends to avoid higher-carbon biomes, reducing ILUC by 30 percent

The baseline of 190 gCO_2e/MJ of ILUC emissions intensity assumed that expansion affected land types more or less randomly. However, because it is anticipated that a disproportionate amount of land expansion will occur in the United States, and not for the most part in the highest-carbon biomes (i.e., farmers will tend to prefer grasslands over forests), the NoSoRFIM assumes that the carbon stock of land converted will be lower than for the global average land parcel. Specifically, the carbon stock of the average parcel of converted land is taken to be only 70 percent of the world average. By effectively curtailing the amount of deforestation associated with land expansion, ILUC emissions intensity is reduced to 30 gCO_2e/MJ.

4.2.8. Relative importance and uncertainty in the factors

The preceding subsections piece together a narrative showing how it is possible at the macroscopic level for ILUC emissions from producing a given biofuel to come in much lower than would be expected if all the feedstock came from land conversion. Of course, in the real world, each of the determinants is a combination of thousands and even millions of individual decisions: farmers deciding to invest in new equipment, companies deciding to clear new areas from their land banks, smallholders choosing whether to clear forests or start cropping areas of pasture, and so forth. As well as providing an illustration of how each determinant affects ILUC, the NoSoRFIM can provide some indication of the importance of each and the uncertainty associated with that significance. In addition to the 'central' case just described, calibrated to the Air Resources Board's ILUC estimate for corn, the model has constructed a best case and a worst case for the ILUC implications of varying each determinant—these are based on judgment and on parameter values reported in the literature. Again, bear in mind that this example is based on corn ethanol. If a different feedstock were being considered, the ranges and magnitudes could be quite different. Figure 4.2 shows the central (30 gCO_2e/MJ), best, and worst cases for ILUC emissions intensity as each factor is changed, as well as what the emissions intensity estimate would be if it were eliminated (e.g., if there were no consideration of co-products or if one were to ignore the likelihood that when U.S. corn demand increases, land use change will be concentrated in the United States). The variation from best case to worst case reflects the importance of each determinant of ILUC emissions but also the level of variation in the treatment of each in existing models and studies.

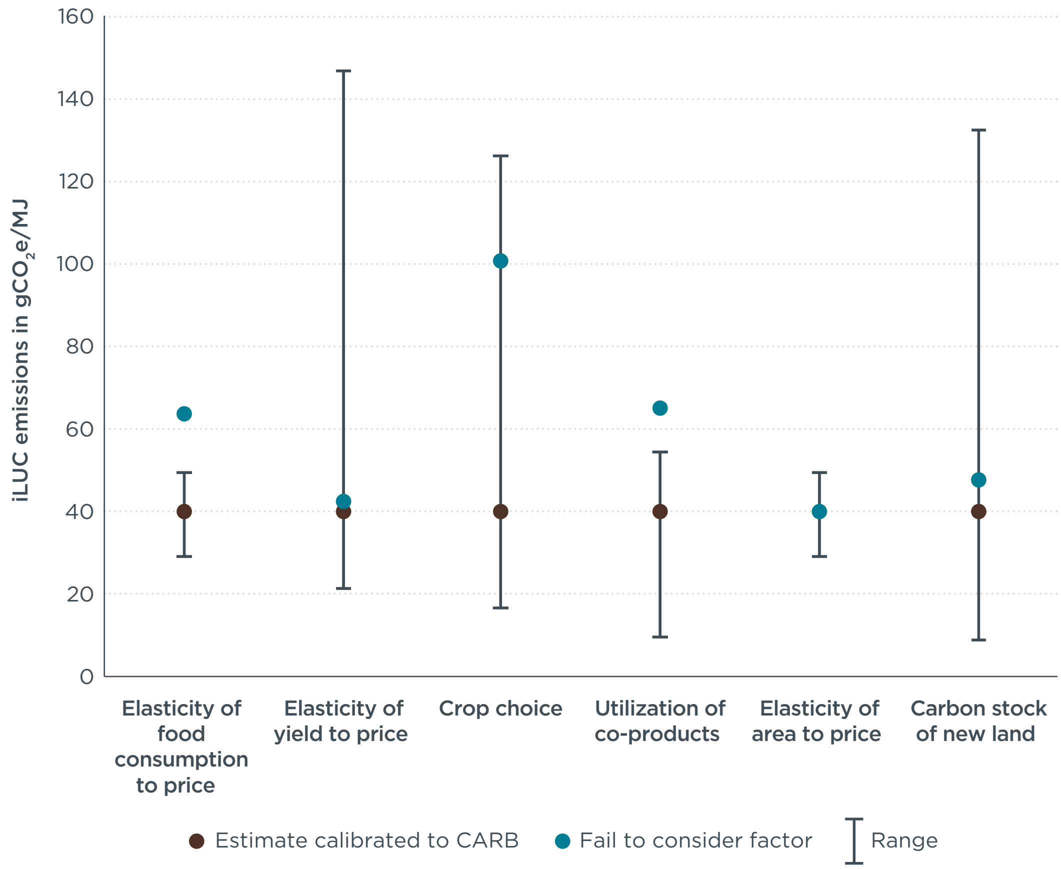

Figure 4.2. Illustrative model of how assumptions for each parameter affect ILUC results for U.S. corn ethanol

Note: The emissions levels marked with blue bars for 'fail to consider' represent the ILUC emissions intensity expected if one were to eliminate a determinant. For instance, ignoring co-products would increase expected emissions intensity by more than 50 percent compared with the central estimate. Because in reality co-products are important and should be considered, this value actually falls outside the range of plausible values.

The largest ranges are those for yield, crop switching, and carbon stock of new land (emissions factors). In the case of yields, this is a result of the literature presenting a wide range of positions. At the worst-case end, ILUC is high if one assumes that there is no direct response of yield to price and that yields on land at the margin of production (the new land used to supply extra feedstock) are much lower than average yields. At the best-case end, the yield response is as strong as the area elasticity response, and new land achieves average productivity. For emissions factors, the range illustrates how much difference it makes whether expansion affects high- or low-carbon ecosystems—in this case, eliminating the parameter means all land use change occurring with an average carbon cost. For crop switching, which has not been widely discussed in the literature, different models have assigned widely varying contributions to its effects.

Where the ranges are narrower, this does not necessarily mean that these determinants are less important. Food consumption reduction, for instance, is essential to reducing ILUC, but there is some degree of consensus in the literature to expect declining food consumption to supply between 20 percent and 50 percent of feedstock (though there are modeling exercises that venture outside this range). The area elasticity has a relatively narrow range assigned to it, but, as noted earlier, this is an effect that runs in parallel to yield change and demand change—so if one were to vary the area elasticity at the same time as the others, a much wider range would ensue.

These ranges serve as a valuable reminder that the outputs from an ILUC model are only as good as the inputs. Using a set of inputs skewed toward optimism would allow a very low (or even negative in some studies) ILUC factor to be reported, whereas using pessimistic parameters could make ILUC emissions look enormous. With the simplified model, combining all the worst-case assumptions produces an ILUC emissions intensity of nearly 3,000 gCO_2e/MJ (this represents a scenario in which extensive land conversion occurs in forest ecosystems at low yields), while combining the best-case assumptions results in a credit (negative emissions intensity) of nearly 30 gCO_2e/MJ (whereby co-products allow land use change to be avoided). The fact that distinct assumptions give different answers should not be treated as surprising, nor should disagreement in the literature be understood as implying that ILUC is so uncertain that model results have no value. The lesson from this exercise is that only by understanding the assumptions that go into building that scenario involving global-scale responses to a biofuel mandate can one judge whether a model gives a reasonable estimate.

4.3. GTAP: COMPUTABLE GENERAL EQUILIBRIUM MODELING FOR U.S. CORN ETHANOL

The ‘walk-through’ of ILUC given above is based on a spreadsheet model built around a six-factor understanding of the processes that mitigate ILUC. As a second illustration of how economic modeling can lay out a scenario for ILUC emissions, the next section of the chapter is devoted to a ‘decomposition’ (breakdown) into the various determinants of the results for ethanol from U.S. corn reported to the European Union’s Joint Research Centre (JRC) by the Global Trade Analysis Project (GTAP) modeling team at Purdue University.[105] The decomposition is based on a modified version of the method reported by Witzke et al. (2010)—for more details see Appendix A. It is important to understand that different decomposition methodologies—for example, the GTAP approach of sequentially activating model features or the JRC (Edwards, Mulligan, and Marelli 2010) approach—will necessarily give slightly discrepant values for the various decomposed elements because they use different units and measure different things.

105 In 2009, the JRC contracted with several economic modeling groups to run indirect land use change scenarios for a marginal increase in biofuel supply of 1 metric megaton of oil equivalent, and it is the results of this modeling for JRC that have been broken down here. The model version used for the JRC is essentially the same as the one used for California’s Air Resources Board, and the team was the same team that undertook the modeling for ARB’s Low Carbon Fuel Standard.

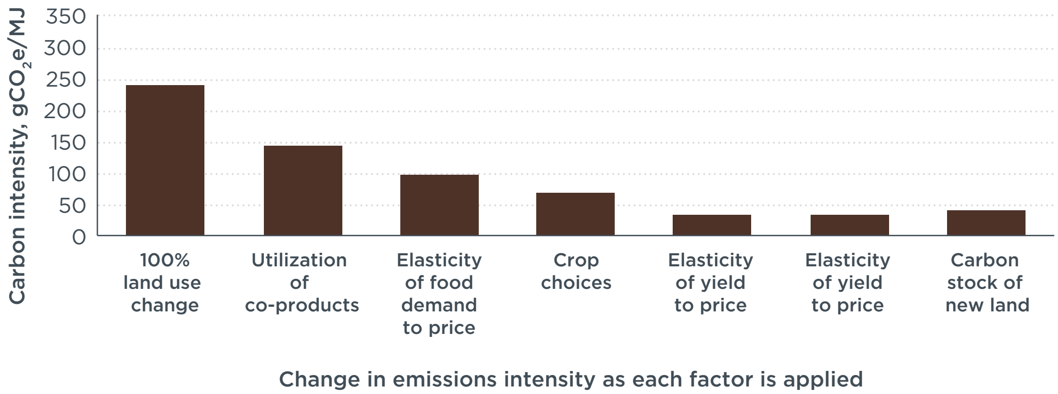

Figure 4.3. Effect of various factors on the emissions reported by GTAP for U.S. corn ethanol

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

The results of this decomposition are displayed graphically in Figure 4.3. The decomposition starts in the leftmost column with emissions for ‘gross land use’—based on the emissions that would result if all production occurred at world average yield using an average emissions factor for the United States (this is slightly different from the use of a world average emissions factor in the NoSoRFIM above). As can be seen, if 100 percent of the increase in biofuels demand were met in this way, through land expansion only, the emissions would be nearly 240 gCO_2e/MJ—much greater than the potential savings from reduced fossil fuel use. However, once all the other determinants have been accounted for, GTAP predicts only 40 gCO_2e/MJ on a European 20-year amortization. That is about 27 gCO_2e/MJ on a 30-year amortization schedule as used in the United States (i.e., close to the LCFS regulatory value). Throughout this section, results are reported based on a 20-year amortization; all emissions intensities can be reduced by a third to give the equivalent 30-year amortized values. Subsequent references to results, predictions, and so forth from GTAP are based on the decomposition analysis as well as any data published by the GTAP modelers themselves.

4.3.1. Elasticity of food demand to price

As feedstock prices increase, elastic consumption means that demand from the food and feed sector falls. GTAP predicts the largest single-country reductions in food and feed consumption to occur in the United States (see Figure 4.4). GTAP predicts ‘sticky’ trade through the Armington framework (see section 3.3.2), so it is not entirely surprising that the strongest price effects, and hence consumption changes, may be felt in the same country/region where the increase in biofuel demand takes place, even though Americans are likely to have less elastic food demand than people in poorer countries.

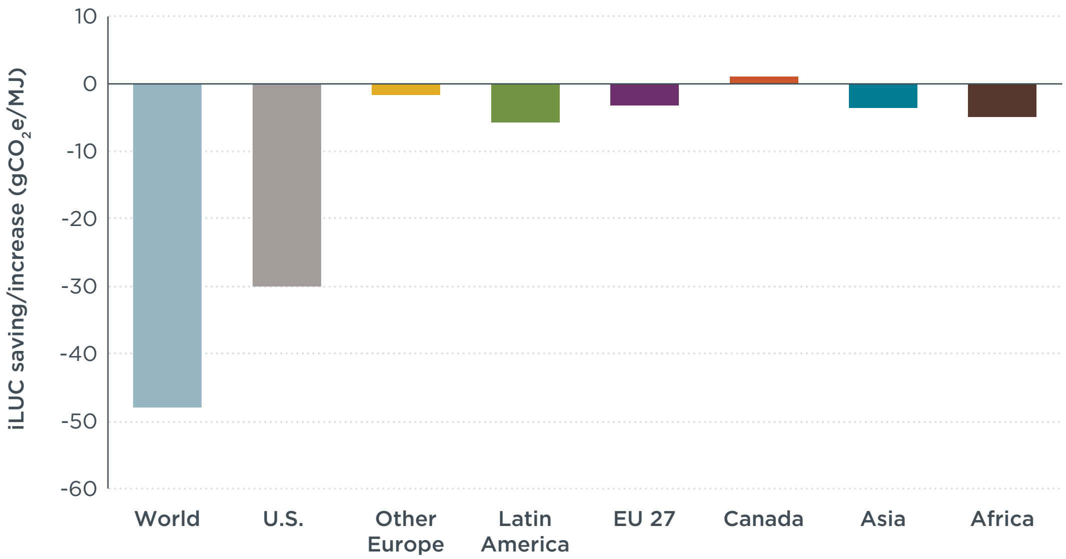

Figure 4.4. ILUC avoided by demand for food and feed by region

Note: ILUC savings are described as negative values; ILUC emissions as positive values.

The demand reduction within the United States is significantly larger than for any other region, accounting for more than half of the total consumption reduction and cutting ILUC emissions intensity by 30 gCO_2e/MJ. The consumption effects are less significant in the regions that one might most associate with food insecurity; there is certainly some degree of food vs. fuel conflict (see section 3.4) in the developing world, but in this modeling it is not a dominant effect. The next most affected regions after the United States are Africa and Latin America, both contributing ILUC reductions of about 5 gCO_2e/MJ. Overall, food and feed consumption reduction reduces gross expected ILUC emissions intensity by about 20 percent, slightly less than 50 gCO_2e/MJ.

4.3.2. Elasticity of yield to price

4.3.2.a. Price-led yield increase

As discussed in section 3.2.3, GTAP models a strong response of yield to price, based on an own-price yield elasticity of 0.2 (Keeney and Hertel 2008). In the decomposition analysis, this effect is important, sparing 15 percent of gross ILUC emissions intensity, or about 35 gCO_2e/MJ. About half of the yield benefit is experienced for corn, primarily in the United States. The rest is achieved mostly for oilseeds and other crops. The gross land demand is reduced by about 13 percent because of this effect. One can see that the balance between supply and demand response varies by region: in Africa, where there is a relatively strong demand response (reduction in food and feed consumption), there are only minor yield gains, while in Asia the yield gains account for almost twice as much ILUC saving as the consumption reduction.

4.3.2.b. Extensive/marginal yield effects

Despite the relatively strong assumption (compared with other models) that yield on new land is only 0.66 times the regional average yield, extensive yield effects are not particularly important in this GTAP scenario, resulting in only 2 gCO_2e/MJ of ILUC savings overall. As noted in section 3.2.6.b, Keeney (2010) shows that the outcomes of GTAP modeling are sensitive to the ratio between the yield on new and average land, so this small effect must hide the aggregate of several different aspects of the modeling that the decomposition is unable to break apart. In particular, there must be an ILUC 'credit' to cancel out the ILUC 'deficit' from expanding onto lower-yielding new land—this credit is probably evidence that, although extension of cultivation happens at a lower than average yield for a given region and crop, that does not necessarily mean lower than average yield worldwide; that is, this value includes elements of crop switching and location choice.

The difficulties in interpreting exactly why this effect appears so small for this scenario highlight the challenge of breaking down complex results, without being able to go into the models themselves and define additional output data specifically for the purpose.

4.3.3. Crop choice

The decomposition analysis defined the gross land demand for this scenario based on world average yields. This is different from Witzke et al. (2010), who used the average yield in the region where the biofuel demand occurred. Starting with the world average yield makes it possible to highlight the importance to the overall ILUC results of where crop expansion is predicted to occur. As expected given the use of the Armington elasticities (section 3.3.2) and the fact that the United States is the world's dominant producer of corn, GTAP predicts that the majority of coarse grains (grains other than wheat or rice) area expansion will indeed happen in the United States. Because U.S. corn yields are much higher than the global average, this concentration of production locally results in a large carbon saving—50 gCO_2e/MJ thanks to the higher yield in the United States. There is also a saving within the U.S. from reducing net exports of various crops. This contributes a further saving of 35 gCO_2e/MJ. These reduced exports/increased imports are seen not only in the corn market (which contributes a saving of 9 gCO_2e/MJ) but in other markets like wheat, soy, and cotton. Overall, these crop location/switching effects save 85 gCO_2e/MJ in the United States (Figure 4.5).

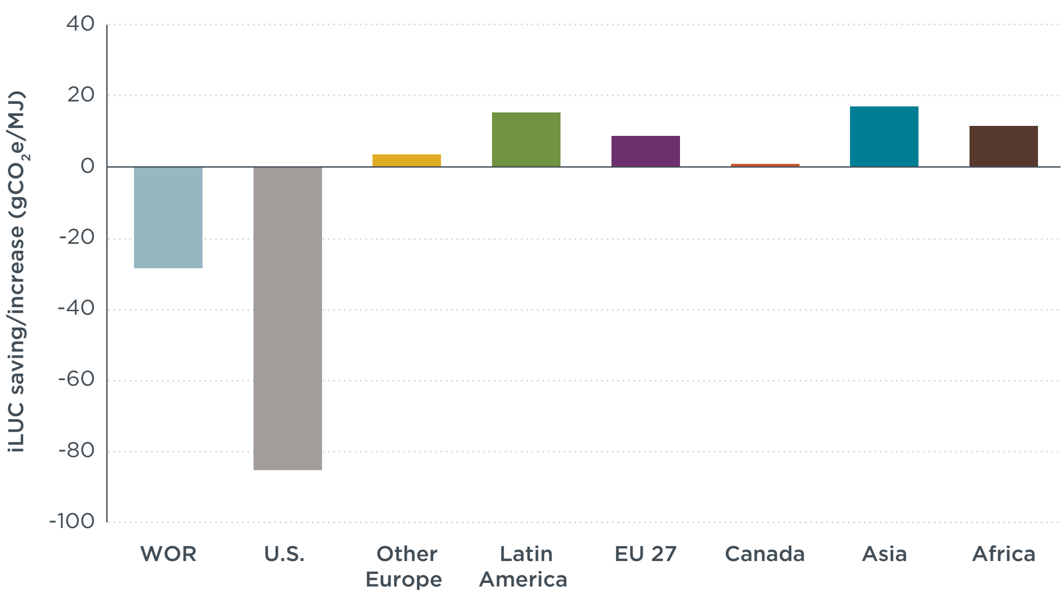

Figure 4.5. ILUC emissions intensity mitigated or aggravated by choice of crop locations and crop switching by region

Note: ILUC savings are described as negative values; ILUC emissions as positive values.

To make up for reduced U.S. exports/increased U.S. imports, there is a production increase in other regions distributed across commodities. For instance, in Asia (Figure 4.6), increasing exports and reducing imports results in increased land requirement and emissions for coarse grains, wheat, oilseeds and other crops (like cotton and tobacco), for a total of about 17 gCO_2e/MJ additional ILUC emissions intensity.

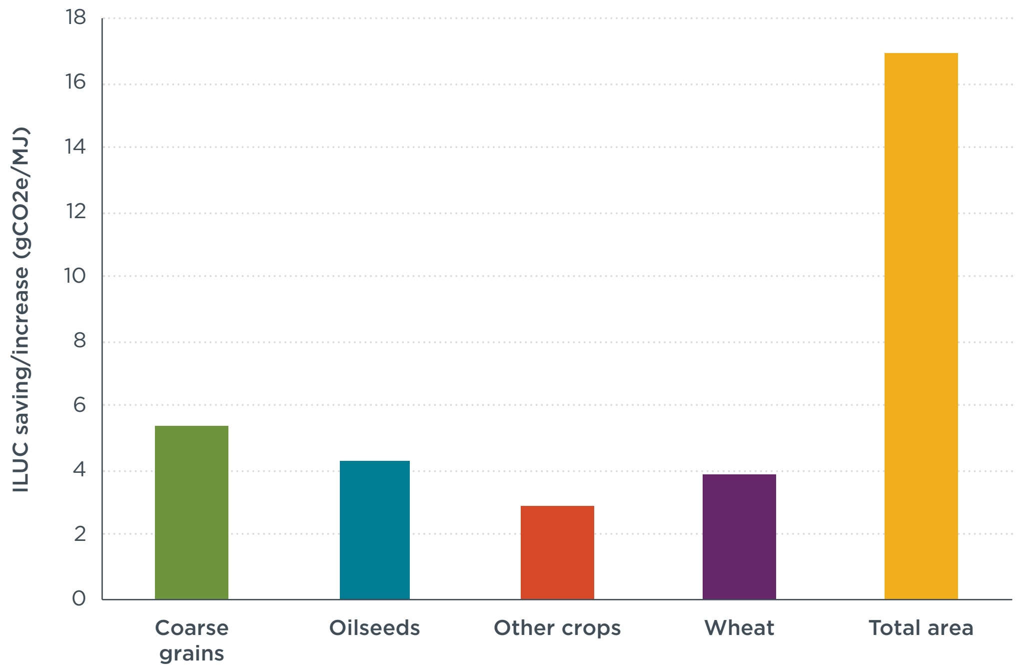

Figure 4.6 ILUC caused by reduced imports to Asia/increased exports from Asia

Overall, the various interactions that contribute to crop location and switching reduce gross ILUC emissions by about 12 percent (see Figure 4.3), with the savings from concentrating corn expansion in the United States more than compensating for the land needed to replace reduced U.S. exports.

4.3.4. Utilization of co-products

GTAP deals with the possible substitution between different commodities that can be used as animal feed via a nest structure, in which commodities at each level can substitute for each other directly. This is illustrated in Figure 4.7 (Golub 2010).

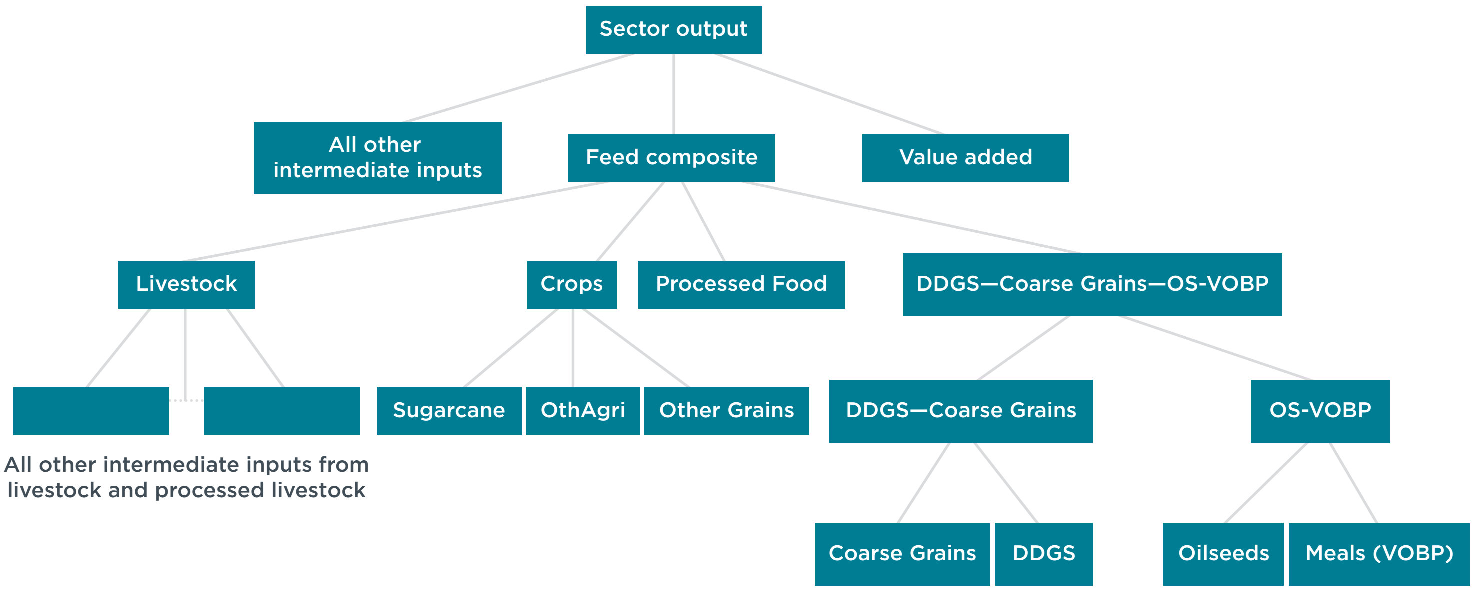

Figure 4.7. Distillers grains and meals in the GTAP product nest structure

At the lowest level of the nest, dried distillers grains (DDGS) sit next to coarse grains (primarily corn), while oil meal (vegetable oil by-product) sits in a nest alongside oilseeds. To understand better how GTAP allows distillers grains to enter the livestock feed market and affect the use of other feed commodities, it is instructive to work through what happens in the nest when the supply of DDGS increases.

First, if supply of DDGS increases, then the price of DDGS will fall. DDGS is now more attractive to livestock owners as a feed, and it will therefore substitute for some of the commodity placed next to it in the nest, coarse grains. With less demand for coarse grains, the price of coarse grains will fall until a new equilibrium with DDGS prices is reached. With both coarse grains and DDGS trading at lower prices than before, the average price of the "DDGS–coarse grains" compound commodity at the next level of the nest will also fall. DDGS–coarse grains now represent a better deal for livestock producers compared with the oilseeds/vegetable oil by-product (OS-VOBP) compound commodity next to them. That compound commodity is made up of oilseeds and oil meals. Therefore, demand for DDGS–coarse grains will increase a little, and demand for oilseeds and oil meals will slacken—and the price of oilseeds and oil meals will both fall slightly. At the next level, the full "DDGS–coarse grains–OS VOBP" commodity has now been slightly reduced in price, so one would also expect to see some amount of substitution of them for other crops, "processed feed" and feed from livestock; however, note that the GTAP documentation makes it fairly clear that substitution at this level is minor compared with the substitution of DDGS for coarse grains at the bottom level of the nest. In the data available to Witzke et al. it is not possible to identify precisely the degree of substitution at the higher levels, so Figure 4.3 is based on assuming that only coarse grains are substituted by DDGS. Because this is the dominant substitution, it should affect the results only slightly.

There are several ways in which the GTAP co-product substitution structure falls short of perfectly modeling the real feed sector. First, in the real feed sector, feeding decisions are made with a view to maintaining the nutritional composition of feed (see section 3.4). DDGS contains energy and protein, as well as other nutritional components, and this will affect the products it substitutes for. GTAP does not include any constraint to maintain the nutritional balance of the feed sector, so it may give outcomes that are inconsistent with real-world practice. Because the primary substitution is for coarse grains (corn), GTAP is unlikely ever to predict oilseeds and meal as the dominant substituted ingredient. However, that is likely to be appropriate for the U.S. market, and the Joint Research Centre (Edwards, Mulligan, and Marelli 2010) writes, "As far as JRC-IE can tell, GTAP realistically models DDGS byproduct from ethanol in the U.S. It replaces both energy-feeds and some oilmeal feed."

In the decomposition illustrated in Figure 4.3, including co-products reduced the gross ILUC emissions by about 40 percent, a saving of nearly 95gCO_2e/MJ. This is similar to the 46 percent reduction in land use requirement reported by GTAP and the result from the JRC's own decomposition methodology (Edwards, Mulligan, and Marelli 2010).

4.3.5. Overall area expansion

Overall, expansion of cultivated area only accounts for a fraction of the feedstock supply for the expanding biofuel market—the amount of area expansion predicted by GTAP for the U.S. corn scenario is just 14 percent, or 32 gCO_2e/MJ, of the gross ILUC if the various factors discussed above are accounted for. Section 3.4 notes that the area expansion is partly determined by the constant elasticity of transformation (CET) functions in GTAP (which calculate the ease with which land changes from one managed use to another). Still, area effects can also in some sense be considered a residual—the amount of feedstock still to be found after yield and consumption changes have been modeled.[106] Figure 4.8 compares the overall magnitude of the different effects (the demand effect from reduced consumption and the two supply effects, yield and area expansion). Globally, the three are of a similar magnitude, with consumption reduction being largest, followed by area expansion, which causes slightly more ILUC emissions than yield increase saves.

This hierarchy of effects can be gauged against the results presented by Roberts and Schlenker (2010) and Berry and Schlenker (2011), which were discussed in section 3.2.3.[107] The GTAP-modeled demand response is broadly comparable to the demand response that Roberts and Schlenker find in the econometric historical data, which is to say, between one-half and one-third of the overall response. The yield response, however, is much stronger in the modeling than in the historical data. The GTAP modelers would argue that the short-run (one-year) response captured by the historical analysis is inevitably underestimated and that the medium- to long-term elasticity of yield to price is much higher, justifying this result. As discussed above, however, this view of yields seems to be somewhat optimistic and only weakly supported by the literature. The yield response in this modeling may be stronger than one would expect in the real world and the area response may be weaker. In other words, on this point, the GTAP results can be seen as offering a lower bound for probable ILUC emissions.

106 In fact, in the model all of these effects must be balanced alongside each other, so each is "co-residual" to the others as a new equilibrium is found.

107 Note that while ILUC emissions scale to some extent with area, and area to some extent with tons of feedstock demand, the inclusion of regional yields and of emissions factors in the decomposition means that one should be cautious when directly comparing results in the units here (gCO_2e/MJ) with results like those in Berry and Schlenker (2011).

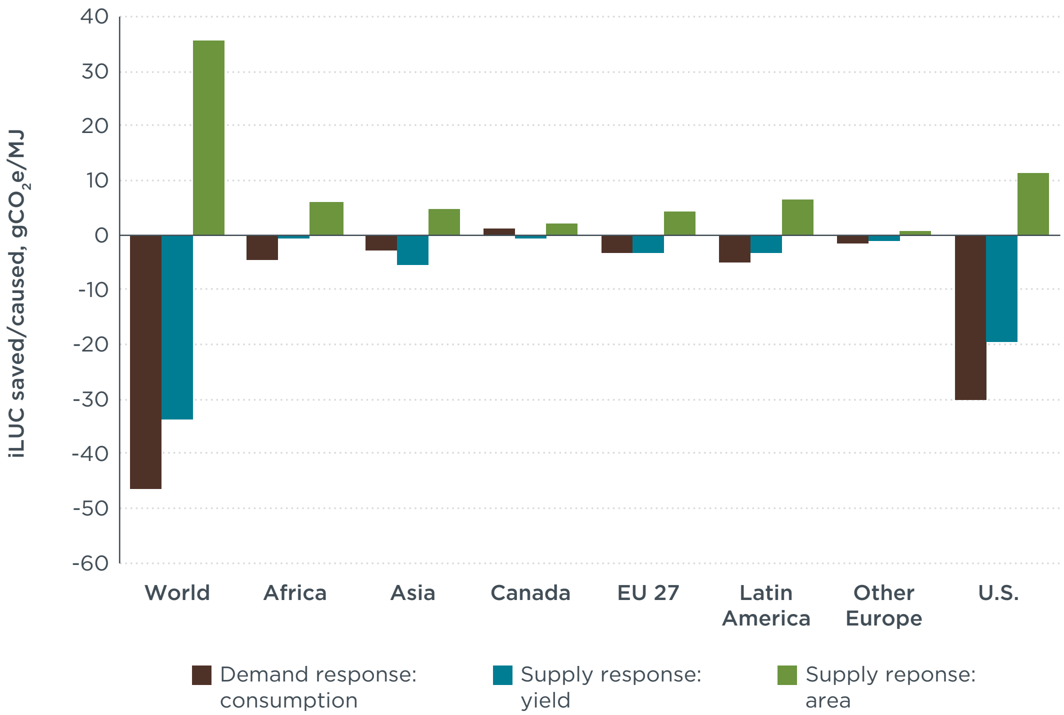

Figure 4.8. Comparing the ILUC avoided and caused by the demand and supply responses

4.3.6. Carbon stock of new land

The final contributor to ILUC emissions from this GTAP modeling is the selection of emissions factors (carbon stock) and the allocation of land use change between high-carbon ecosystems (forest) and (relatively) low-carbon ecosystems (pasture). Using a set of regional average emissions factors taken from the literature, but ignoring the different land types GTAP allocates to conversion, one would find a total ILUC emissions intensity of 32 gCO_2e/MJ from the land use change predicted by this GTAP scenario. This is somewhat below the emissions intensity estimated by GTAP itself for this run, of 40 gCO_2e/MJ. It is important to remember that GTAP's emissions allocation model is more detailed than the one applied in the decomposition. GTAP's associated emissions factor model explicitly accounts for the ratio of forest to pasture conversion, distinguishes agro-ecological zones, etc., all of which can help to explain the variance. The disparity in calculated emissions may also be attributable to differences in source data; the dataset GTAP used for this scenario (from the Woods Hole Oceanographic Institution) estimates higher-carbon stocks for some types of land cover than the dataset used in the decomposition—see Section 3.6.2 for more detail. Clearly, emissions factors and the determination of the type of land that has been converted play an important role in determining whether ILUC emissions are high or low.

5. CONCLUSIONS

Toward the beginning of this report, it was noted that if all of the feedstock for biofuels policy (European wheat for ethanol in the example in section 1.1.1) came from land expansion, then even if the land converted had relatively low carbon stocks, the emissions would likely still cancel out any carbon savings from reduced fossil fuel use.[108] In order to be able to assert that a biofuel support policy will reduce carbon emissions, any biofuel advocate must be able to propose and justify a scenario in which biofuel feedstock demand is met either through cultivation of land that would otherwise have persistently low carbon stocks or else not just from new land but also through yield improvement and demand reduction. To put it another way, given that biofuel use does not reduce emissions from the tailpipes of vehicles, claims about environmental benefits must be based on identifying where in the global agricultural system net carbon sequestration will increase as a result of a biofuels policy.

Most often, commentators who believe that indirect land use change emissions from some or all biofuels are low[109] (or that there is no competition with food) tend to point to one of the following responses:

- » Yield increases;
- » Co-product utilization.

In addition to these two factors, three further ways that ILUC emissions can be avoided or offset are identifiable:

- » Food consumption reduction;
- » Expansion on land with very low carbon release on conversion;
- » Crop switching, either replacing one crop type with another or shifting crops between regions.

This report has continually reminded the reader that one cannot consider any one of these responses on its own—the determinative question is whether other changes will outpace the expansion of cultivated land in response to higher prices, or whether the land expansion response will be the dominant one, or whether they are of comparable magnitude. If the elasticity of agricultural area to price is high compared with other possible responses, and some studies imply it is, then it will only be by carefully targeting land use choices rather than allowing them to be made by the market that ILUC can be kept to a level that retains greenhouse gas benefits from biofuel mandates.

It is only by analyzing the expectations for each of these responses together—as has been done widely using economic modeling tools—that one can reasonably conclude that a biofuel policy is a good greenhouse gas mitigation policy.

108 One exception might be perennial crops, in cases where they could actually increase rather than reduce sequestration of soil carbon.

109 In addition to the more considered positions, there are various simplistic arguments made that ILUC is nonexistent or negligible. The simplest is outright denial of the concept, while others are built around statistics that might seem prima facie incompatible with ILUC—an often-quoted example is that Amazon deforestation has decelerated since 2000 while biofuel production has increased. Such arguments often fail to pose the baseline question properly and conflate correlation with causation (in this case, the question is not whether deforestation rates are lower than in 2000 but whether they are lower than they would have been now without biofuel policies). There is also a common fallacy on both sides of the dialogue of treating ILUC as only a problem of faraway, ultra-high-carbon ecosystems like tropical rainforests. In fact, conversion of grasslands and even pastures still has carbon consequences.

5.1. ELASTICITY OF FOOD DEMAND TO PRICE

'Food versus fuel' is arguably the most controversial aspect of the ongoing debate about the merits of biofuel policies. Nobody on any side of the discussion wants to achieve carbon reductions primarily by reducing the food consumption of people who are vulnerable to hunger. Supporters of biofuels have often denied that there will be any impact at all, while some critics of biofuels in the development community have invoked emotional imagery of malnourished multitudes.

The basic economics of food versus fuel is clear: increases in demand on the scale represented by biofuels policies will cause food price increases, and price increases lead to diminished consumption. It seems fair to expect that the poor[110] will disproportionately experience these impacts, but, even so, in absolute terms (metric tons) the bulk of reduced food consumption may well occur in the developed countries where the mandates are issued rather than in developing countries. In the literature on the 2007/2008 food price spikes, there is broad consensus that biofuels were an important cause, though not necessarily a dominant one. Claims that biofuels were not involved are based either on misunderstanding of the facts or on a refusal to accept the available analysis. The modeling literature consistently predicts that food prices will rise as biofuel targets increase—and that the welfare losses because of this will be greater overall than any benefits to farmers and producers.[111] Several studies predict that global expansion of biofuel demand will push tens of millions beneath the poverty line and will increase global inequality even when potential economic benefits are taken into account.

The flip side of this competition between food and fuel is that as consumption falls, so does the demand for additional land for cultivation. The consumption demand response might reasonably range between 50 percent and 100 percent of the size of the combined supply response from land expansion and yield improvements. The results of economic modeling tend to be broadly consistent with that expectation. The combination of a large reduction in food consumption and a strong yield response (implying a weak land expansion response) could result in a slashing of net land demand by 75 percent[112]—the sort of land savings that make low ILUC factors plausible.

One key area for further research is the impact of biofuels policy on food price volatility and whether there are policy options available to moderate it. The dynamics of short-term price changes are more complex than the underlying supply/demand balance that drives steadier trends, and it may well be that the largest impact on welfare of biofuels policies is not medium-term price increases but the amplification of short-term food crises. On the other hand, there may be opportunities, as discussed by Laborde (2011b), to use biofuels policy to damp price fluctuations, with positive implications for food security and welfare.

110 As well as reduced nutritional intake, this will lead to welfare losses through reduced disposable income.

111 Although several modelers (e.g., Laborde 2011a) conclude that these welfare losses are relatively minor compared with the scale of the global economy, Wiggins and Levy (2008) suggest that such losses could be compensated for by additional policy measures.

112 If the supply response were equal to the combined demand response (the top of the range in Roberts and Schlenker [2010]) and the yield response matched the area response (outside the range in Berry and Schlenker [2011] but potentially consistent with arguments that yield is more responsive than in the past).

5.2. ELASTICITY OF YIELD TO PRICE

Section 3.2 explored the response of agricultural yield to price and demand. It revealed that the literature documents the lack of any compelling evidence that, historically, yields have reacted strongly to price. As well, it found that analysis of the year-on-year response to price changes suggests on balance that agricultural area is more (and perhaps much more) responsive to price than yield is. There was evidence that yields on new land following expansion of cultivation were likely to be lower than on existing land, although the size of the difference is not yet clear.

Nevertheless, a fairly robust yield response has been included in most economic modeling of ILUC, and it is generally assumed to provide an amount of feedstock comparable to the extra feedstock from land expansion. Given the assumptions about yield response compared with area response in models like the Global Trade Analysis Project (GTAP), the International Food Policy Research Institute's Modeling International Relationships in Applied General Equilibrium (IFRPI-MIRAGE), and the Food and Agricultural Policy Research Institute (FAPRI), it seems unlikely that the yield response to biofuel policies is being underestimated; hence, criticism of the models for allegedly underestimating yield responses is probably not well founded.

On a related note, there is a degree of consensus in the literature on agricultural yield, as well as on the proposition that public investment in agriculture is an important driver of yield improvement. Regardless of biofuels policy, it would be appropriate for governments to consider improving agricultural productivity through this type of direct investment rather than aiming to do it at two removes via the channel of biofuels-induced price increases. Despite the focus by some commentators on yield increase as a source of additional biofuel feedstock, yields are unlikely to increase so much in response to biofuel demand that they supply the bulk of the additional feedstock required. For ILUC emissions from biofuels policies to be low would require contributions from other sources; yields alone will not be enough.

Two questions about yield remain contentious in ILUC modeling, and achieving more clarity regarding them would be of great value in improving confidence in ILUC estimates. First, additional econometric and microeconomic research to explore the responsiveness of yield to prices would help pin down a narrower range for yield to price elasticities. Second, further work to measure directly yields being achieved by farms expanding at the margin of production, especially in the developed world, would help confirm, calibrate, or refute the approaches to estimating marginal yield currently used by the models.

5.3. CO-PRODUCT UTILIZATION

There is no doubt that the availability of co-products reduces the requirement for new land for biofuels production. Ignoring this fact would result in a substantial overestimate of likely ILUC emissions. But the magnitude of the ILUC savings from co-products depends on expectations about interactions in the livestock feed market.

For co-products used as animal feed, it seems generally likely that the demand for land will be reduced by roughly the proportion of the dry mass of a crop that ends up in co-products, so if one-third of the dry mass of corn goes into distillers grains, then the net demand for extra corn would be only two-thirds of the gross demand.

For some feedstocks and co-products, however, there may be an additional credit to be had, if the co-product displaces a different crop that is more strongly linked to land use change emissions. The example of this that has been most explored is that of wheat distillers grains from European ethanol production cutting imports of Brazilian soy meal, with the notion that Brazilian soy is more closely tied to deforestation and land use change emissions than European wheat is. It is indeed likely that wheat distillers grains will displace some quantity of soy meal, and therefore insofar as there is evidence that soy cultivation aggravates deforestation, the argument has considerable merit. That said, there are complications that need to be considered. Some economic modeling suggests that, owing to other price effects, the net change in soy imports to Europe under a biofuel mandate may not be that large, and there may be a further layer of emissions consequences if soybean oil production drops and must be replaced by palm oil. Still, it certainly seems possible that this effect could reduce ILUC substantially for EU wheat ethanol, and such links should be considered when analyzing other crops.

To date, the best data available on co-product use tend to be based on feed trials and study of linear programs designed for feed optimization. As the ethanol industries in the United States and the European Union mature, there are more and more data available on the actual utilization patterns of distillers grains and oil meal in livestock feed markets. It would be extremely valuable to investigate these market data to provide confirmation and calibration of predictions about feed displacement.

5.4. CROP CHOICE

Crop location is important because some regions have much more efficient cropping systems than others. If U.S. corn ethanol growth caused expansion in the United States rather than in Africa, this would help minimize land use requirements. Whether biofuels expansion will happen in the same regions where the policies are introduced depends largely on how interconnected agricultural trade is (and how well increased U.S./EU demand translates into higher world market prices). If markets are less tightly integrated, one might expect lower ILUC emissions. Conversely, if agricultural commodities are effectively part of a single world market, one might tend to expect higher ILUC emissions (although, as always, this will depend on specific crop choices and the interaction with emissions factors).

Crop shifting is something potentially important based on the results of the economic ILUC models, yet it is not typically invoked by biofuel advocates as a source of ILUC reductions, and it is not included in causal descriptive modeling such as that done by the consultancy E4tech (2010). Crop shifting entails crop choices changing globally in a way that makes the system as a whole more efficient at sequestering carbon. For example, if high-yielding oilseeds replace lower-yielding oilseeds, one might 'gain' land at no carbon cost. This seems to be one of the effects that make the ILUC factors predicted by MIRAGE lower. It is important to ask whether there are any other implications of such crop shifting. For instance, another MIRAGE prediction points to a decrease in fruit and vegetable consumption (to be replaced by other food sources), which might be considered a negative outcome.

While it seems plausible that there might be room for these sorts of efficiency gains in the agricultural system, crop-shifting effects have not been thoroughly discussed, or often even recognized, in the ILUC literature. It seems equally plausible that the models where crop shifting is important may be anticipating transitions that are unrealistic for reasons not captured by the models: there are presumably good reasons why farmers would be growing lower-yielding crops. Such predictions ought to be carefully sense-checked before they are used as a basis for biofuels policies promising to deliver carbon savings.

Crop switching is, to some extent, a catchall for a variety of choices available in the agricultural economy. Despite the importance of this set of outcomes in some of the ILUC models, crop choices have not been systematically investigated in order to validate the model results. Additional research to document thoroughly what various models actually predict and to compare those predictions with available real-world data would have great value in calibrating future model runs.

5.5. ELASTICITY OF AREA TO PRICE

There would be no ILUC if agricultural area were not responsive to price and hence to demand. Unlike yield, where it has been difficult to find clear econometric evidence of a historical link to price, several studies have found strong evidence for changes in cultivated area. As many nations move away from government control of agriculture to market-led decision making, it seems plausible that cultivated area will be more responsive to price movements and demand. Still, it is difficult to identify parameter values clearly for use in models. Al-Riffai, Dimaranan, and Laborde (2010b, p. 92) claimed that "there are no robust estimates from the econometric literature." The models currently rely heavily on data from the developed world, which could introduce a bias into expectations about area response, but it is unclear which way this bias would go.

The most important question regarding area elasticity to price, though, is not its absolute parameter value but how it compares to yield and food demand elasticities. In ILUC modeling terms, it would be preferable to have all the price elasticities wrong in the same way, leading to a pronounced bias, than to have the food demand and yield elasticity calculated well but have the area elasticity off by an order of magnitude.

Ensuring that the area elasticity is well calibrated against the other elasticities is a key task for modelers and remains an important area for further study. Additional research on how the elasticity of expansion compares for different land types would also be invaluable. Further work contrasting changes in cropped areas, grassland areas, and forest areas would be of great assistance to future modeling efforts.

5.6. CARBON STOCK OF NEW LAND

Section 3.6 considered the carbon stock of newly cultivated land and the associated emissions factors. It showed that best estimates suggest that even categories of land often considered appropriate for biofuel expansion (such as low-biodiversity grasslands or abandoned agricultural land) contain large stocks of carbon relative to any potential carbon savings associated with expanding in those areas. On average, most ecosystems have increasing carbon sequestration year-on-year, so it is proper to consider the carbon cost of forgone sequestration in assessing the benefits of biofuel policies—this is especially true of abandoned agricultural lands in naturally heavily forested regions like Northern Europe. While there are opportunities to plant crops with high-carbon sequestration to replace ecosystems with low-carbon sequestration (a good example is the possibility of replacing degraded *Imperata* grassland in Indonesia with oil palms; use of perennial grasses as energy crops would also offer advantages), these are the exception rather than the rule. Section 3.6 also mentioned that there are some ecosystems with extremely high carbon stocks, notably, peatlands in Malaysia and Indonesia but also rain forests, other forests, and wetlands elsewhere. Even a relatively small amount of expansion into these systems could cancel out the potential benefits of biofuels.

Without specific measures to ensure that agricultural expansion across the entire sector is directed onto land that is not only economically but ecologically most suitable, the average carbon cost of land expansion for biofuel production is bound to be significant. The International Energy Agency (IEA 2011) recognizes that growth in biofuels cannot be sustainable unless land use change is more actively controlled. It recommends that the world "adopt an overall sustainable land-use management system that aims to ensure all agricultural and forestry land is comprehensively managed in a balanced manner to avoid negative indirect land-use change and support the wide range of demands in different sectors." Such recommendations are, however, unlikely to be implemented any time in the near future.

One of the major recurring questions about land use emissions is whether there is a stock of low-carbon abandoned or fallow land that is being (and will continue to be) turned over to agriculture in response to biofuel demand. Tyner et al. (2010), for instance, introduce a "cropland pasture" category to the GTAP model and argue that this is likely to come under cultivation more readily than forested land. Further work reviewing the historical evidence on the extent to which such expectations are justified would be instructive in resolving this ongoing issue in the modeling.

5.7. IS THERE A CORRECT ANSWER?

This report has laid out an introduction to a broad field of study that covers questions from pig nutrition to agricultural economics to peat degradation. It has identified six key factors that determine ILUC emissions. Each of these represents a sum of choices being made every day by people all over the world: how much food to buy; whether to invest in a new tractor; which crops to plant; what to feed to the cows; whether to plant an extra field or leave it fallow; whether to protect or exploit forests. The effect of each of these determinants, whether in real life or in any given model, depends on a complex web of interactions. Modeling them requires many assumptions to be made, and the available data do not yet give clear answers for all parameters.

As with any attempt to predict outcomes and pose counterfactuals, there are uncertainties that can be resolved through additional evidence, but also some that cannot be 'solved' because the future is always uncertain. As many commentators have noted, the uncertainty is not about whether ILUC happens. There will inevitably be some land use changes driven by biofuels policy, and these changes will surely have emissions consequences. The uncertainty is about what magnitude of atmospheric carbon benefits biofuels will deliver, indeed, whether they will deliver any carbon benefits at all.

There are clearly several pathways to low ILUC emissions. Most will involve a combination of assumptions about vigorous yield growth in response to higher prices, standard to optimistic assumptions on food demand adjusting to accommodate biofuels feedstock supply, and assumptions that emissions factors will be toward the low end of the range—typically coupled with an added ILUC credit from co-products or crop shifting or both. There are also clearly many scenarios in which the cultivated land expansion response causes enough ILUC emissions either to eliminate or severely reduce any CO_2 benefits from biofuel mandates. In the context of the European ILUC discussion (Marelli, Mulligan, and Edwards 2011; Malins 2011a), caution was given that, if ILUC emissions are not dealt with, biofuels policies are likely to be an exceedingly expensive way to deal with climate change, in terms of dollars or euros per metric ton of carbon dioxide abatement (if they deliver any benefit at all). It seems unlikely that taxpayers and consumers will indefinitely support policies that represent a substantial wealth transfer from them to agricultural producers and biofuel processors, unless they can be given confidence that the policies are delivering the promised benefits at a reasonable cost.

Recognizing the importance of ILUC casts a question mark over the carbon savings that have previously been assumed for biofuels and makes them less attractive as a policy option to mitigate climate change. Nevertheless, there is a positive side to the results of ILUC modeling because 'reverse-engineering' the logic of ILUC points to opportunities to reduce emissions from biofuels and from agriculture in general. As in the best-case scenarios presented in section 4.1, it is possible to direct a strategy whereby ILUC can be avoided through crop switching, co-product utilization, and low emissions factors. The paper "Biofuels Done Right" by Dale et al. (2010) shows (for the case of the U.S. corn ethanol mandate) that if the response to biofuel demand could be limited to the United States only, with beneficial crop switching, cultivated area expansion targeted onto idle land with relatively low-carbon stocks, and the introduction of innovations in livestock nutrition to allow cellulosic biofuel co-products to be used as feed, then a large-scale U.S. biofuels industry could be possible without major ILUC emissions or impacts on food production. Market-based biofuel mandates alone would not be enough to make this scenario happen. As Dale and colleagues point out, "Multiple drivers would be required to actually produce these changes." Even so, there is an opportunity for policymakers to work with the farming industry to introduce a new generation of more sophisticated regulatory guidance to help allow this positive model of development to unfold.

The existing generation of models and results provides a basis on which to build intelligent policies, informed by uncertainty rather than confounded by it. The U.S. Environmental Protection Agency and California's Air Resources Board have already contended with ILUC by quantifying and regulating it, using FAPRI-FASOM and GTAP respectively. These models give the agencies an evidentiary basis to judge which biofuel pathways are most likely to help reduce net carbon emissions. The incorporation of ILUC into these regulations has provided a clear opportunity in the market for biofuels that do not consume more land and has given an added impetus to the development of fuels from wastes, cellulose, algae, and so forth. In Europe, the European Commission has used MIRAGE to model ILUC, and various European institutions aim to agree on an approach to deal with land use change during the coming year. This could include ILUC factors calculated in a manner similar to the California system, and it seems likely to involve capping the amount of food-based biofuels eligible for government support. In each of these jurisdictions, policymakers will need to respond as new evidence emerges and as the world changes in unpredictable ways (imagine the impact on land use change expectations of a global treaty on deforestation).

One limitation of ILUC factors as a basis to identify preferred biofuel pathways is that in some cases low ILUC emissions may be correlated with larger impacts on food markets. In the worst case, ILUC factors could systematically favor feedstocks that have the highest potential to harm social welfare. Given the chronic problems of food insecurity and malnutrition in the developing world, it is appropriate to act with caution when it comes to policies that intervene on this scale in food markets. One approach mooted by the Air Resources Board would be to disallow credit in ILUC calculations for reduction in food consumption, effectively a carbon penalty proportional to the effect a biofuel is expected to have on food supplies.

An alternative to reliance on ILUC factors would be to design policies explicitly targeted to encourage the types of agricultural practice that could reduce or avoid these risks. A focus on avoiding ILUC was recommended by the UK government's Gallagher Review (UK RFA 2008), and recognized approaches to bypassing ILUC would be to concentrate more heavily on wastes and residues;[113] to look for opportunities for 'additionality of production'[114] as in the Responsible Cultivation Areas system developed by Ecofys (2009); to pursue the development of nonland resources like algae; and to adopt agricultural models that utilize degraded, marginal, or otherwise underutilized and low-carbon land.

In response to the broad consensus that biofuels both raise food prices and heighten price volatility, there have been calls for biofuels support to be modified, moderated, or abandoned (e.g., FAO et al. 2011). If those calls are to be taken seriously without abandoning the hoped-for benefits from biofuels, it might be appropriate to transition from the ambitious volume targets in place for existing models of production, toward a more cautious and narrower set of measures designed to foster investment in new, low-ILUC, welfare-positive production models. Measures that allow farmers to generate additional value for residues and to develop profitable cropping systems on land unsuited to traditional crops might enhance rural development without the negative welfare impact that has been tied to the current raft of policies.

There are good biofuels, but there is a burden of proof on both government and industry to provide convincing evidence that biofuel mandates as a policy intervention can be reasonably expected to deliver the carbon savings that are claimed for them. For the current generation of biofuels, this evidence has to include ILUC modeling, and the evidence on ILUC is compelling that indiscriminate biofuel mandates are unlikely to realize carbon reduction policy objectives. More broadly, this assessment suggests that a change is warranted in international conventions for biofuel carbon accounting. For example, the Kyoto Protocol and the European Emissions Trading System treat biofuels as 'carbon neutral.' Unless both direct and indirect emissions are recognized, the incentives offered to biofuels climate policies will not be properly aligned to the benefits on offer.

113 Some wastes are more truly discarded than others. For wastes or residues that already have uses in the economy, it may be unproductive in net carbon terms to divert supplies to biofuel production.

114 In the Ecofys methodology, production is considered additional if it would not have occurred without investment directly linked to biofuel production.

6. REFERENCES

Adams, D., R. Alig, B. A. McCarl., and B. C. Murray (2005). *FASOMGHG Conceptual Structure, and Specification: Documentation.* Texas A&M University. February. Available online at http://agecon2.tamu.edu/people/faculty/mccarl-bruce/papers/1212FASOMGHG_doc.pdf

ADAS (UK) (2008). *Anticipated and Potential Improvements in Land Productivity and Increased Agricultural Inputs with Intensification A Study Comissioned by AEA Technology as Part of the Gallagher Biofuels Review for Renewable Fuels Agency, Department for Transport.* Version 3.4. May 21. Available online at http://www.globalbioenergy.org/uploads/media/0805_ADAS_-_Anticipated_and_potential_improvements_in_land_productivity_and_increased_agricultural_inputs_with_intensification.pdf

Ahmed, S. A., T. W. Hertel, and R. Lubowski (2008). "Calibration of a Land Cover Supply Function Using Transition Probabilities." GTAP Research Memorandum no. 14. Center for Global Trade Analysis, Purdue University. October. Available online at https://www.gtap.agecon.purdue.edu/resources/download/4199.pdf

Air Resources Board, California (ARB), Low Carbon Fuel Standard Expert Workgroup on Indirect Land Use Change (2010). Reports available online at http://www.arb.ca.gov/fuels/lcfs/workgroups/ewg/expertworkgroup.htm

Air Resources Board, California (ARB), Expert Workgroup Investigating Indirect Effects of Transportation Fuels (2010). *Issues Related to Accounting for Co-Product Credits in the California Low Carbon Fuel Standard.* White paper. December 8. Available online at http://www.arb.ca.gov/fuels/lcfs/workgroups/ewg/010511-final-rpt-coproduct-credits.pdf

Air Resources Board. (2009). Low Carbon Fuel Standard: Final Regulation Order. Available at: http://www.arb.ca.gov/fuels/lcfs/CleanFinalRegOrder112612.pdf

Air Resources Board, California (ARB), Low Carbon Fuel Standard Expert Workgroup (2011). "Final Recommendations from the Elasticity Values Subgroup." Available online at http://www.arb.ca.gov/fuels/lcfs/workgroups/ewg/010511-final-rpt-elasticity.pdf

Air Resources Board, California (ARB) (2012), Carbon Intensity Lookup Table for Gasoline and Fuels that Substitute for Gasoline (as of December 2012). Available online at http://www.arb.ca.gov/fuels/lcfs/lu_tables_11282012.pdf

Akhurst, M., N. Kalas, and J. Woods (2011). "Applying Causal Descriptive Modelling Techniques to the Assessment of the Potential Indirect Land Use Change Impact of French Grown Oilseed Rape Biodiesel." LCAworks, Imperial College London.

Al-Riffai, P., B. Dimaranan, and D. Laborde (2010a). *European Union and United States Biofuel Mandates: Impacts on World Markets.* Technical Notes no. IDB-TN-191. Inter-American Development Bank. December. Available online at http://www.globalbioenergy.org/uploads/media/1012_IDB_-_European_Union_and_United_States_biofuel_mandates.pdf

Al-Riffai, P., B. Dimaranan, and D. Laborde (2010b). *Global Trade and Environmental Impact Study of the EU Biofuels Mandate.* Final report.International Food Policy Research Institute, on behalf of the European Commission Directorate-General for Trade. March. Available online at http://trade.ec.europa.eu/doclib/docs/2010/march/tradoc_145954.pdf

Alexandratos, N. (1999). "World Food and Agriculture: Outlook for the Medium and Longer Term." *Proceedings of the National Academy of Sciences* 96, no. 11 (May 25): 5908–14.

Anderson, J. L., D. J. Schingoethe, K. F. Kalscheur, and A. R. Hippen (2006). "Evaluation of Dried and Wet Distillers Grains Included at Two Concentrations in the Diets of Lactating Dairy Cows." *Journal of Dairy Science* 89, no. 8 (August): 3133–42.

Arima, E. Y., P. Richards, R. Walker, and M. M. Caldas (2011). "Statistical Confirmation of Indirect Land Use Change in the Brazilian Amazon." *Environmental Research Letters* 6, no. 2 (April–June): 024010.

Armington, P. S. (1969). "A Theory of Demand for Products Distinguished by Place of Production." *Staff Papers—International Monetary Fund* 16, no.1 (March): 159–78.

Arora, S., M. Wu, and M. Wang (2008). "Update of Distillers Grains Displacement Ratios for Corn Ethanol Life-Cycle Analysis." Report no. ANL/ESD/11-1. Center for Transportation Research, Energy System Division, Argonne National Laboratory, U.S. Department of Energy. September. Available online at http://greet.es.anl.gov/publication-3bi0z09m

Ash, M. S., and W. Lin (1987). "*Regional Crop Yield Response for U.S. Grains.*" Agricultural Economic Report no. 577. Economic Research Service, U.S. Department of Agriculture. Available online at http://naldc.nal.usda.gov/download/CAT10853172/PDF

Babcock, B. A., and M. Carriquiry (2010). "An Exploration of Certain Aspects of CARB's Approach to Modeling Indirect Land Use from Expanded Biodiesel Production." Staff Report no. 10-SR 105. Center for Agricultural and Rural Development, Iowa State University. February. Available online at http://www.card.iastate.edu/publications/dbs/pdffiles/10sr105.pdf

Baffes, J., and T. Haniotis (2010). "Placing the 2006/08 Commodity Price Boom into Perspective." Policy Research Working Paper no. 5371. Development Prospects Group, World Bank. July. Available online at http://www-wds.worldbank.org/servlet/WDSContentServer/WDSP/IB/2010/07/21/000158349_20100721110120/Rendered/PDF/WPS5371.pdf

Balmford, A., R. E. Green, and J. P. W. Scharlemann (2005). "Sparing Land for Nature: Exploring the Potential Impact of Changes in Agricultural Yield on the Area Needed for Crop Production." *Global Change Biology* 11, no. 10 (October): 1594–1605.

Banse, M., H. van Meijl, A. Tabeau, and G. Woltjer (2008). "Will EU Biofuel Policies Affect Global Agricultural Markets?" *European Review of Agricultural Economics* 35, no. 2 (June): 117–41.

Barr, K., B. Babcock, M. Carriquiry, A, Nasser and L. Harfuch (2010). "Agricultural Land Elasticities in the United States and Brazil", Food and Agricultural Policy Research Institute (FAPRI) Publications, Food and Agricultural Policy Research Institute (FAPRI) at Iowa State University.

Bell, M. A., R. A. Fischer, D. Byerlee, and K. Sayre (1995). "Genetic and Agronomic Contributions to Yield Gains: A Case Study for Wheat." *Field Crops Research* 44, nos.2–3 (December): 55–65.

Berry, S. T. (2011). "Biofuels Policy and the Empirical Inputs to GTAP Models." Yale University Department of Economics and Cowles Foundation and National Bureau of Economic Research. January 4. Available online at http://www.arb.ca.gov/fuels/lcfs/workgroups/ewg/010511-berry-rpt.pdf

Berry, S., and W. Schlenker (2011). "Technical Report for the ICCT: Empirical Evidence on Crop Yield Elasticities." International Council on Clean Transportation. August 5. Available online at http://www.theicct.org/empirical-evidence-crop-elasticities

Bouët, A., L. Curran, B. Dimaranan, M. P. Ramos, and H. Valin (2009). “Biofuels: Global Trade and Environmental Impact Study.” Final Report submitted to the European Commission. April 29. (This report is not publicly available to the best of knowledge; this report has quoted it via EC DG Energy [2010].)

Brander, M., C. Hutchison, C. Sherrington, A. Ballinger, C. Beswick, A. Baddeley, M. Black, J. Woods, and R. Murphy (2009). “Methodology and Evidence Base on the Indirect Greenhouse Gas Effects of Using Wastes, Residues, and By-products for Biofuels and Bioenergy.” Report to the Renewable Fuels Agency and the Department for Energy and Climate Change by Ecometrica, Eunomia Research & Consulting, and Imperial College of London. November 30. Available online at http://www.bioenergywiki.net/images/a/a5/Ecometrica_Methodology.pdf

Brisson, N., P. Gate, D. Gouache, G. Charmet, F.-X. Oury, and F. Huard (2010). “Why Are Wheat Yields Stagnating in Europe? A Comprehensive Data Analysis for France.” *Field Crops Research* 119, no. 1 (October 9): 201–12.

Britz, W., and T. W. Hertel (2011). “Impacts of EU Biofuels Directives on Global Markets and EU Environmental Quality: An Integrated PE, Global CGE Analysis.” *Agriculture, Ecosystems and Environment* 142, nos. 1–2 (July): 102–9.

Bruinsma, J. (2009). “The Resource Outlook to 2050: By How Much Do Land, Water and Crop Yields Need to Increase by 2050?” Paper presented at the FAO Expert Meeting on How to Feed the World in 2050. Food and Agriculture Organization of the United Nations. Rome. June 24–26. Available online at ftp://ftp.fao.org/docrep/fao/012/ak971e/ak971e00.pdf

Calderini, D. F., and G. A. Slafer (1998). “Changes in Yield and Yield Stability in Wheat during the 20th Century.” *Field Crops Research* 57, no. 3 (June): 335–47.

Cassman, K. G. (1999). “Ecological Intensification of Cereal Production Systems: Yield Potential, Soil Quality, and Precision Agriculture.” *Proceedings of the National Academy of Sciences* 96, no. 11 (May 25): 5952–59.

Casson, A. L. Tacconi, and K. Deddy (2007). “Strategies to Reduce Carbon Emissions from the Oil Palm Sector in Indonesia.” Paper prepared for the Indonesian Forest Climate Alliance in advance of the United Nations Framework Convention on Climate Change Conference. Bali. December.

Chadwick, D. R., B. F. Pain, S. K. E. Brookmann (2000). “Nitrous Oxide and Methane Emissions Following Application of Animal Manures to Grassland.” *Journal of Environmental Quality* 29, no. 1 (January–February): 277–87.

Ciampitti, I. A., E. A. Ciarlo, and M.E. Conti (2008). “Nitrous Oxide Emissions from Soil during Soybean (*Glycine max* [L.] Merrill) Crop Phenological Stages and Stubbles Decomposition Period.” *Biology and Fertility of Soils* 44, no. 4 (March): 581–88.

Cororaton, C. B., G. Timilsina, and S. Mevel (2010). “Impacts of Large Scale Expansion of Biofuels on Global Poverty and Income Distribution”, Contributed Paper at the IATRC Public Trade Policy Research and Analysis Symposium “Climate Change in World Agriculture: Mitigation, Adaptation, Trade and Food Security” June 27 - 29, 2010 Universität Hohenheim, Stuttgart, Germany.

Croezen, H., and F. Brouwer (2008). “Estimating Indirect Land Use Impacts from By-Products Utilization.” CE Delft. June. Available online at http://www.ce.nl/art/uploads/file/08_4723_30.pdf

Dalal, R. C., W. Wang, G. P. Robertson, and W.J. Parton (2003). "Nitrous Oxide Emission from Australian Agricultural Lands and Mitigation Options: A Review." *Australian Journal of Soil Research* 41, no. 2 (April): 165–95.

Dale, B. E., B. D. Bals, S. Kim, and P. Eranki. "Biofuels Done Right: Animal Feeds Enable Large Environmental and Energy Benefits." *Environmental Science and Technology* 44, no. 22 (November 15): 8385–89.

Dale, V. H., K. L. Kline, J. Wiens, and J. Fargione (2010). "Biofuels: Implications for Land Use and Biodiversity." Biofuels and Sustainability Reports. Ecological Society of America. January. Available online at http://www.esa.org/biofuelsreports/files/ESA%20Biofuels%20Report_VH%20Dale%20et%20al.pdf

Davidson, E. A., and I. L. Ackerman (1993). "Changes in Soil Carbon Inventories Following Cultivation of Previously Untilled Soils." *Biogeochemistry* 20, no. 3 (September): 161–93.

De Hoyos, R. E., and D. Medvedev (2009). "Poverty Effects of Higher Food Prices: A Global Perspective." (March 1, 2009). World Bank Policy Research Working Paperno. 4887. March 1. Available online at http://ssrn.com/abstract=1372964

Delaney, M., S. Brown, A. E. Lugo, A. Torres-Lazama, and N. Bello Quintero (1998). "The Quantity and Turnover of Dead Wood in Permanent Forest Plots in Six Life Zones of Venezuela." *Biotropica* 30, no. 1 (March): 2–11.

Demeyer, D. I., and C. J. Van Nevel (1975). "Methanogenesis, an Integrated Part of Carbohydrate Fermentation, and its Control." In *Digestion and Metabolism in the Ruminant. Proceedings of the IV International Symposium on Ruminant Physiology, Sydney, Australia, August 1974.* Ed. I. W. McDonald and A. C. I. Warner. Armidale, Australia: University of New England Publishing Unit. Pp. 366–82.

Devarajan, S., and S. Robinson (2002). "The Influence Of Computable General Equilibrium Models On Policy." TMD Discussion Paper no. 98. Trade and Macroeconomics Division, International Food Policy Research Institute. August. Available online at http://www.unc.edu/depts/econ/seminars/tmdp98.pdf

Don, A., J. Schumacher, and A. Freibauer (2011). "Impact of Tropical Land-Use Change on Soil Organic Carbon Stocks—A Meta-Analysis." *Global Change Biology* 17, no. 4 (April): 1658–70.

Du, X., and D. J. Hayes (2008). "The Impact of Ethanol Production on U.S. and Regional Gasoline Prices and on the Profitability of the U.S. Oil Refinery Industry." Working Paper no. 08-WP 467. Center for Agricultural and Rural Development, Iowa State University. April. Available online at http://www.card.iastate.edu/publications/dbs/pdffiles/08wp467.pdf

Dumortier, J., and D. J. Hayes (2009). "Towards an Integrated Global Agricultural Greenhouse Gas Model: Greenhouse Gases from Agriculture Simulation Model (GreenAgSiM)." Working Paper no. 09-WP 490. Center for Agricultural and Rural Development, Iowa State University. May. Available online at http://www.card.iastate.edu/publications/dbs/pdffiles/09wp490.pdf

Dumortier, J,, D. J. Hayes, M. Carriquiry, F. Dong, X. Du, A. Elobeid, J. F. Fabiosa, and S. Tokgoz (2009). "Sensitivity of Carbon Emission Estimates from Indirect Land-Use Change." Working Paper no. 09-WP 493.Center for Agricultural and Rural Development, Iowa State University. July. Available online at http://www.card.iastate.edu/publications/dbs/pdffiles/09wp493.pdf

Dumortier, J., D. J. Hayes, M. Carriquiry, F. Dong, X. Du, A. Elobeid, J. F. Fabiosa, and K. Mulik (2010). "Modeling the Effects of Pasture Expansion on Emissions from Land-Use Change." Working Paper no. 10-WP 504 (revised). Center for Agricultural and Rural Development, Iowa State University. September. Available online at http://www.card.iastate.edu/publications/dbs/pdffiles/10wp504.pdf

Dunn, J. B., S. Mueller, H.-Y. Kwon, and M. Q. Wang (2013). "Land-Use Change and Greenhouse Gas Emissions from Corn and Cellulosic Ethanol." *Biotechnology for Biofuels* 6, no. 1 (April): 51.

Dyson, T. (1999). "World Food Trends and Prospects to 2025." *Proceedings of the National Academy of Sciences* 96, no. 11 (May 25): 5929–36.

E4tech (2010). *A Causal Descriptive Approach to Modelling the GHG Emissions Associated with the Indirect Land Use Impacts of Biofuels.* Final report. A study for the UK Department of Transport. October. Available online at http://www.apere.org/doc/1010_e4tech.pdf

Earles, J. M., S. Yeh, and K. E. Skog (2012). "Timing of carbon emissions from global forest clearance." Nature Climate Change, 2(9), 682-685.

Ecofys (2009). "A Review "of Demand-Yield Relationships." Draft. August 31.

Ecofys (2010). "Responsible Cultivation Areas: Identification and Certification of Feedstock Production with a Low Risk of Indirect Effects." September. Available online at http://www.ecofys.com/files/files/ecofysrcamethodologyv1.0.pdf

Ecofys (2011). "Indirect Effects of Biofuel Production." Overview prepared for the Global Bioenergy Parnership (GBEP). May 18. Available online at http://www.cbd.int/agriculture/2011-121/eu-ecofys2-sep11-en.pdf

Edwards, R., D. Mulligan, and L. Marelli (2010). *Indirect Land Use Change from Increased Biofuels Demand: Comparison of Models and Results for Marginal Biofuels Production from Different Feedstocks.* JRC Scientific and Technical Reports no. EUR 24485 EN – 2010. Institute for Energy, Joint Research Centre, European Commission. Available online at http://ec.europa.eu/energy/renewables/studies/doc/land_use_change/study_4_iluc_modelling_comparison.pdf

Edwards, R., J.-F. Larivé, and J.-C. Beziat (2011). *Well-to-Wheels Analysis of Future Automotive Fuels and Powertrains in the European Context: WELL-toTANK Report.* Version 3c. JRC Scientific and Technical Reports no. EUR 24952 EN – 2011. Institute for Energy and Transport, Joint Research Centre, European Commission, together with CONCAWE and Renault/EUCAR. Available online at http://optiresource.org/pdf/JRC_Eucar/wtw3_wtt_report_eurformat.pdf

Ellis, S., J. Webb, T. Misselbrook, and D. Chadwick (2001). "Emission of Ammonia (NH3), Nitrous Oxide (N_2O) and Methane (CH_4) from a Dairy Hardstanding in the UK." *Nutrient Cycling in Agroecosystems* 60, nos. 1–3 (July): 115–22.

Elobeid, A., M. Carriquiry, J. F. Fabiosa, K. Mulik, D. J. Hayes, B. A. Babcock, J. Dumortier, and F. Rosas (2011). "Greenhouse Gas and Nitrogen Fertilizer Scenarios for U.S. Agriculture and Global Biofuels." Working Paper no. 11-WP 524. Center for Agricultural and Rural Development, Iowa State University. June.

EMPA (Swiss Federal Laboratories for Materials Science and Technology) (2012). *Harmonisation and Extension of the Bioenergy Inventories and Assessment.* End report. Commissioned by the Bundesamt für Energie (BFE). August 31.

Ericksen, M. H., and K. Collins (1985). "Effectiveness of Acreage Reduction Programs." In *Agricultural–Food Policy Review: Commodity Programs Perspectives.* Agricultural Economic Report no. 530. Economic Research Service, U.S. Department of Agriculture. July.

Ernst & Young (2011). *Biofuels and Indirect Land Use Change: The Case for Mitigation.* Report commissioned by the European Renewable Ethanol Association, the International Union for the Conservation of Nature, Neste Oil, Partners for Euro-African Green Energy, Riverstone Holdings, and Shell. October. Available online at http://www.endseurope.com/docs/111005a.pdf

Escobar, L. F., T. J. C. Amado, C. Bayer, L. F. Chavez, J. A. Zanatta, and J. E. Fiorin (2010). "Postharvest Nitrous Oxide Emissions from a Subtropical Oxisol as Influenced by Summer Crop Residues and their Management." *Revista Brasileira de Ciência do Solo* 34, no. 2 (March–April): 435–42.

European Commission, Directorate-General for Energy (EC DG Energy) (2010). *The Impact of Land Use Change on Greenhouse Gas Emissions from Biofuels and Bioliquids: Literature Review.* July. Available online at http://ec.europa.eu/energy/renewables/studies/doc/land_use_change/study_3_land_use_change_literature_review_final_30_7_10.pdf

Evans, L. T. (1997). "Adapting and Improving Crops: The Endless Task." *Philosophical Transactions of the Royal Society—Series B: Biological Sciences* 352, no. 1356 (July 29): 901–6.

Fargione, J., J. Hill, D. Tilman, S. Polasky, and P. Hawthorne (2008). "Land Clearing and the Biofuel Carbon Debt." *Science* 319, no. 5867 (February 29): 1235–38.

Financial Times (2008). "Biofuel for Thought." Editorial. July 8. Available online at http://www.ft.com/intl/cms/s/0/05bed884-4d30-11dd-b527-000077b07658.html

Finger, R. (2010). "Evidence of Slowing Yield Growth—The Example of Swiss Cereal Yields." *Food Policy* 35, no. 2 (April): 175–82.

Food and Agriculture Organization of the United Nations (FAO) (1980). *Global Forest Resources Assessment.* Available online at http://www.fao.org/forestry/fra/24691/en/

Food and Agricuture Organization of the United Nations (FAO) (2008). *The State of Food and Agriculture—Biofuels: Prospects, Risks and Opportunities.* Available online at ftp://ftp.fao.org/docrep/fao/011/i0100e/i0100e.pdf

Food and Agriculture Organization of the United Nations, Committee on World Food Security, High Level Panel of Experts on Food Security and Nutrition (FAO HLPE) (2011). *Price Volatility and Food Security.* HLPE Report no. 1. July. Available online at http://www.fao.org/fileadmin/user_upload/hlpe/hlpe_documents/HLPE-price-volatility-and-food-security-report-July-2011.pdf

Food and Agriculture Organization of the United Nations, Committee on World Food Security, High Level Panel of Experts on Food Security and Nutrition (FAO HLPE) (2013). Biofuels and food security. HLPE Report no. 5. June. Available online at http://www.fao.org/fileadmin/user_upload/hlpe/hlpe_documents/HLPE_Reports/HLPE-Report-5_Biofuels_and_food_security.pdf

Food and Agriculture Organization of the United Nations (FAO), International Fund for Agricultural Development, International Monetary Fund, Organisation for Economic Co-operation and Development, United Nations Conference on Trade and Development, United Nations World Food Programme, World Bank, World Trade Organization, International Food Policy Research Institute, and the UN Secretary-General's High-Level Task Force on the Global Food Security Crisis (2011). *Price Volatility in Food and Agricultural Markets: Policy Responses.* Policy Report to the G20 group of nations. June 2. Available online at http://www.worldbank.org/foodcrisis/pdf/Interagency_Report_to_the_G20_on_Food_Price_Volatility.pdf

Fritsche, U. (2010). *The "iLUC Factor" as a Means to Hedge Risks of GHG Emissions from Indirect Land Use Change.* Working paper. Öko-Institut. July. Available online at http://www.oeko.de/oekodoc/1030/2010-082-en.pdf

Galbally, I. E., P. J. Fraser, C. P. Meyer, and D. W. T. Griffith (1992). "Biosphere-Atmosphere Exchange of Trace Gases over Australia." In: *Australia's Renewable Resources: Sustainability and Global Change.* Ed. R. M. Gifford and M. M. Barson. Parkes, Australia: Bureau of Rural Resources and CSIRO Division of Plant Industry. Pp. 117–49.

Gilbert, C. L. (2010). "How to Understand High Food Prices." *Journal of Agricultural Economics* 61, no. 2 (June): 398–425.

Godfray, H. C. J., J. R. Beddington, I. R. Crute, L. Haddad, D. Lawrence, J. F. Muir, J. Pretty, S. Robinson, S. M. Thomas, and C. Toulmin (2010). "Food Security: The Challenge of Feeding 9 Billion People." *Science* 327, no. 5967 (February 12): 812–18.

Golub, A. (2010). "Calculation of the Effects of Increased Demand for Biofuel Feedstock on theWorld Agricultural Markets: Intermediate Results and Characteristics of the GTAP Model." Paper presented at the JRC Workshop on the Effects of Increased Demand for Biofuel Feedstocks on the World Agricultural Markets and Areas. Joint Research Centre, European Commission. Ispra, Italy. February 10–11. Available online at http://iet.jrc.ec.europa.eu/sites/bf-ca/files/files/documents/Ispra_v2_GOLUB.pdf

Golub, A., T. W. Hertel, H.-L. Lee, and N. Ramankutty (2006). *Modeling Land Supply and Demand in the Long Run.* Report prepared for the Ninth Annual Conference on Global Economic Analysis. Addis Ababa. June 15–17. Available online at https://www.gtap.agecon.purdue.edu/resources/download/2745.pdf

Guo, L. B., and R. M. Gifford (2002). "Soil Carbon Stocks and Land Use Change: A Meta Analysis." *Global Change Biology* 8, no. 4 (April): 345–60.

Gurgel, A., J. M. Reilly, and S. Paltsev (2007). "Potential Land Use Implications of a Global Biofuels Industry." *Journal of Agricultural and Food Industrial Organization* 5, no 2 (December): 1–36 (Article 9 in the issue "Explorations in Biofuels Economics, Policy, and History").

Hafner, S. (2003). "Trends in Maize, Rice, and Wheat Yields for 188 Nations over the Past 40 Years: A Prevalence of Linear Growth." *Agriculture, Ecosystems and Environment* 97, nos. 1–3 (July): 275–83.

Harris, N., S. Grimland, and S. Brown (2008). "GHG Emission Factors for Different Land-Use Transitions in Selected Countries of the World." Report submitted to U.S. Environmental Protection Agency. Winrock International.

Harris, N., S. Grimland, and S. Brown (2009). "Land Use Change and Emission Factors: Updates since the RFS Proposed Rule."Report submitted to the U.S. Enivronmental Protection Agency. Winrock International.

Hay, R. K. M. (1995). "Harvest Index: A Review of Its Use in Plant Breeding and Crop Physiology." *Annals of Applied Biology* 126, no. 1 (February): 197–216.

Hazzledine, M., A. Pine, I. Mackinson, J. Ratcliffe, and L. Salmon (2011). "Estimating Displacement Ratios of Wheat DDGS in Animal Feed Rations in Great Britain." Working Paper no. 2011-8. International Council on Clean Transportation. November. Available online at http://www.theicct.org/estimating-displacement-ratios-wheat-ddgs-animal-feed-rations-great-britain

Hertel, T. W., A. A. Golub, A. D. Jones, M. O'Hare, R. J. Plevin, and D. M. Kammen (2010a). "Effects of US Maize Ethanol on Global Land Use and Greenhouse Gas Emissions: Estimating Market-Mediated Responses." *BioScience* 60, no. 3 (March): 223–31.

Hertel, T. W., A. A. Golub, A. D. Jones, M. O'Hare, R. J. Plevin, and D. M. Kammen (2010b). Supporting online materials for "Effects of US Maize Ethanol on Global Land Use and Greenhouse Gas Emissions: Estimating Market-Mediated Responses." *BioScience* 60, no. 3 (March): 223–31.

Hertel, T. W., K. Steigert, and H. Vroomen (1996). "Nitrogen-Land Substitution in Corn Production: A Reconciliation of Aggregate and Firm-Level Evidence." *American Journal of Agricultural Economics* 78, no. 1 (February): 30–40.

Hiederer, R., F. Ramos, C. Capitani, R. Koeble, V. Blujdea, O. Gomez, D. Mulligan, and L. Marelli (2010). *Biofuels: A New Methodology to Estimate GHG Emissions from Global Land Use Change: A Methodology Involving Spatial Allocation of Agricultural Land Demand and Estimation of CO_2 and N_2O Emissions.* JRC Scientific and Technical Reports no. EUR 24483 EN – 2010. Joint Research Centre, European Commission. Available online at http://eusoils.jrc.ec.europa.eu/esdb_archive/eusoils_docs/other/EUR24483.pdf

Hindrichsen, I. K., H.–R. Wettstein, A. Machmüller, C. R. Soliva, K. E. Bach Knudsen, J. Madsen, and M. Kreuzer (2004). "Effects of Feed Carbohydrates with Contrasting Properties on Rumen Fermentation and Methane Release in vitro." *Canadian Journal of Animal Science* 84, no. 2 (June): 265–76.

Hoag, D. L., B. A., Babcock, and W. E. Foster (1993). "Field-Level Measurement of Land Productivity and Program Slippage." *American Journal of Agricultural Economics* 75, no. 1 (February): 181–89.

Hoffman, L. A., and A. Baker (2011). *Estimating the Substitution of Distillers' Grains for Corn and Soybean Meal in the U.S. Feed Complex.* Report no. FDS-11-I-01. Economic Research Service, U.S. Department of Agriculture. October. Available online at http://www.ers.usda.gov/media/236568/fds11i01_2_.pdf

Holt, M. T. (1999). "A Linear Approximate Acreage Allocation Model." *Journal of Agricultural and Resource Economics* 24, no. 2 (December):383–97.

Holter, P. (1997). "Methane Emissions from Danish Cattle Dung Pats in the Field." *Soil Biology and Biochemistry* 29, no. 1 (January): 31–37.

Houck, J.P., and M.E. Ryan (1972). "Supply Analysis for Corn in the United States: The Impact of Changing Government Programs." *American Journal of Agricultural Economics* 54, no. 2 (May): 184–91.

Huang, H., and M. Khanna (2010). "An Econometric Analysis of U.S. Crop Yield and Cropland Acreage: Implications for the Impact of Climate Change." Paper presented at the Agricultural and Applied Economics Association 2010 AAEA, CAES, and WAEA joint annual meeting. Denver. July 25–27. Available online at http://ageconsearch.umn.edu/bitstream/61527/2/Econometric%20estimation%20on%20acreage_Submitted.pdf

Huang, Y., J. Zuo, X. Zheng, Y. Wang, and X. Xu (2004). "Nitrous Oxide Emissions as Influenced by Amendment of Plant Residues with Different C:N Ratios." *Soil Biology and Biochemistry* 36, no. 6 (June): 973–81.

International Energy Agency (IEA) (2010). *Medium-Term Oil and Gas Markets 2010.* OECD/IEA. Available online at http://www.iea.org/publications/freepublications/publication/mtogm2010.pdf

International Energy Agency (IEA) (2011). *Technology Roadmap: Biofuels for Transport.* OECD/IEA. Available online at http://www.iea.org/publications/freepublications/publication/biofuels_roadmap.pdf

International Monetary Fund (2010). *International Financial Statistics: Yearbook 2010.* Vol. LXIII. Washington, D.C.: IMF Statistics Department.

IPCC (2006a). *IPCC Guidelines for National Greenhouse Gas Inventories. Vol. 4: Agriculture, Forestry and Other Land Use.* Chapter 4: Forest Land. Available online at http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_04_Ch4_Forest_Land.pdf

IPCC (2006b). *IPCC Guidelines for National Greenhouse Gas Inventories. Vol. 4: Agriculture, Forestry and Other Land Use.* Chapter 2: Generic Methodologies Applicable to Multiple Land-Use Categories. Available online at http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_02_Ch2_Generic.pdf

IPCC (2006c). *IPCC Guidelines for National Greenhouse Gas Inventories. Vol. 4: Agriculture, Forestry and Other Land Use.* Chapter 6: Grassland. Available online at http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_06_Ch6_Grassland.pdf

IPCC (2007). Revised 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Available online at: http://www.ipcc-nggip.iges.or.jp/public/2006gl/

Jacobs Consultancy (2009). *Life Cycle Assessment Comparison of North American and Imported Crudes.* Report prepared for the Alberta Energy Research Institute.

Kebreab, E., K. Clark, C. Wagner-Riddle, and J. France (2006). "Methane and Nitrous Oxide Emissions from Canadian Animal Agriculture: A Review." *Canadian Journal of Animal Science* 86, no. 2 (June): 135–57.

Keeney, R. (2010). "Yield Response and Biofuels: Issues and Evidence on the Extensive Margin." Paper presented at the Fourth World Congress of Environmental and Resource Economists. Montreal. June 28–July 2.

Keeney, R., and T. W. Hertel (2008). "The Indirect Land Use Impacts of U.S. Biofuel Policies: The Importance of Acreage, Yield, and Bilateral Trade Responses." GTAP Working Paper no. 52. Revised version. Center for Global Trade Analysis, Purdue University. October 22. Available online at https://www.gtap.agecon.purdue.edu/resources/download/4104.pdf

Keppler, F., J. T. G. Hamilton, M. Braß, and T. Röckmann (2006). "Methane Emissions from Terrestrial Plants under Aerobic Conditions." *Nature* 439, no. 7073 (January 12): 187–91.

Kim, S., and B. E. Dale (2005). "Environmental Aspects of Ethanol Derived from No-Tilled Corn Grain: Nonrenewable Energy Consumption and Greenhouse Gas Emissions." *Biomass and Bioenergy* 28, no. 5 (May): 475–89.

Kim, S., and B. E. Dale (2011). "Indirect Land Use Change for Biofuels: Testing Predictions and Improving Analytical Methodologies." *Biomass and Bioenergy* 35, no. 7 (July): 3235–40.

Klasing, K. C. (2012). “Displacement Values for US Corn DDGS—Informed by Regionally Specific Least-Cost Diet Formulation for All Major Livestock Types.” Working Paper no. 2012-3. International Council on Clean Transportation. May. Available online at http://www.theicct.org/displacement-ratios-us-corn-ddgs

Klopfenstein, T. J., G. E. Erickson, and V. R. Bremer (2008). Board-Invited Review: “Use of Distillers By-Products in the Beef Cattle Feeding Industry.” *Journal of Animal Science* 86, no. 5 (May): 1223–31.

Kucharik, C. J., and N. Ramankutty (2005). “Trends and Variability in U.S. Corn Yields over the Twentieth Century.” *Earth Interactions* 9, no. 1 (March): 1–29.

Laborde, D. (2011a). *Assessing the Land Use Change Consequences of European Biofuel Policies.* Final Report. Report undertaken by the International Food Policy Research Institute for the Directorate-General for Trade, European Commission. October. Available online at http://trade.ec.europa.eu/doclib/docs/2011/october/tradoc_148289.pdf

Laborde, D. (2011b). “Domestic Policies in a Globalized World: What You Do is What I Get—Consequences of Biofuel Mandates for Global Price Stability. Revised version, Food Security Portal, International Food Policy Research Institute. April 25. Available online at http://www.foodsecurityportal.org/sites/default/files/A_brief_overview_of_Foodsecurity_and_Biofuels_1.pdf

Laborde, D., and H. Valin (2012). “Modeling Land-Use Changes In A Global Cge: Assessing The EU Biofuel Mandates With The MIRAGE-Biof Model.” Climate Change Economics, 3(03).

Le Mer, J., and P. Roger (2001). “Production, Oxidation, Emission and Consumption of Methane by Soils: A Review.” *European Journal of Soil Biology* 37, no. 1 (January–March): 25–50.

Lin, M., and P. Huybers (2012). “Reckoning Wheat Yield Trends.” *Environmental Research Letters* 7, no. 2 (June): 024016.

Litton, C.M., M. G. Ryan, and D. H. Knight (2004). “Effects of tree density and stand age on carbon allocation patterns in postfire lodgepole pine.” Ecological Applications, 14(2), 460-475.

Liu, Y., and C. R. Shumway (2007). “Demand and Supply of Induced Innovation: An Application to U.S. Agriculture.” Paper presented at the American Agricultural Economics Association annual meeting. Portland, Oregon. July 29–August 1. Available online at http://ageconsearch.umn.edu/bitstream/9844/1/sp07li01.pdf

Lobell, D. B., W. Schlenker, and J. Costa-Roberts (2011). “Climate Trends and Global Crop Production since 1980.” *Science* 333, no. 6042 (July 29): 616–20.

Love, H. A., and W. E. Foster (1990). “Commodity Program Slippage Rates for Corn and Wheat.” *Western Journal of Agricultural Economics* 15, no. 2 (December): 272–81.

Lumpkins, B.S., A. B. Batal and N. M. Dale (2005). “Use of distillers dried grains plus solubles in laying hen diets.” The Journal of Applied Poultry Research , 14, 25-31.

Luo, Z., E. Wang, and O. J. Sun (2010). “Soil Carbon Change and its Response to Agricultural Practices in Australian Agro-Ecosystems: A Review and Synthesis.” *Geoderma* 155, nos. 3–4 (March 15): 211–23.

Lywood, W., J. Pinkney, and S. Cockerill (2009). “Impact of Protein Concentrate Coproducts on Net Land Requirement for European Biofuel Production.” *GCB Bioenergy* 1, no. 5 (October): 346–59.

MacKenzie, A. F., M. X. Fan, and F. Cadrin (1997). "Nitrous Oxide Emission as Affected by Tillage, Corn-Soybean-Alfalfa Rotations and Nitrogen Fertilization." *Canadian Journal of Soil Science* 77, no. 2 (May): 145–52.

Maimonides, M. (12th century). *The Guide for the Perplexed.*

Malhi, Y. (2010). "The Carbon Balance of Tropical Forest Regions, 1990–2005." *Current Opinion in Environmental Sustainability* 2, no. 4 (October): 237–44.

Malins, C. (2011a). "Indirect Land Use Change in Europe—Considering the Policy Options: Assessing the 2011 IFPRI MIRAGE iLUC Modelling for the European Commission and Possible Policy Alternatives." Working paper no. 2011-9. International Council on Clean Transportation. November. Available online at http://www.theicct.org/sites/default/files/publications/ICCT_iLUC-EU-options_Nov2011-1.pdf

Malins, C. (2011b). "IFPRI-MIRAGE 2011 Modelling of Indirect Land Use Change: Briefing on Report for the European Commission Directorate General for Trade."Briefing. International Council on Clean Transportation. November. Available online at http://www.theicct.org/sites/default/files/publications/ICCT_IFPRI-iLUC-briefing_Nov2011-1.pdf

Malins, C. (2013). "Vegetable Oil Markets and the EU Biofuel Mandate." Briefing. International Council on Clean Transportation. February. Available online at http://www.theicct.org/sites/default/files/publications/ICCT_vegoil_and_EU_biofuel_mandate_20130211.pdf

Malthus, T. R. (1798). "An Essay on the Principle of Population as it Affects the Future Improvement of Society, with Remarks on the Speculations of Mr. Godwin, M. Condorcet, and Other Writers." London: Printed for J. Johnson, in St. Paul's Church-yard.

Marelli, L., D. Mulligan, and R. Edwards (2011). Critical Issues in Estimating ILUC Emissions: Outcomes of an Expert Consultation, 9–10 November 2010, Ispra (Italy). JRC Scientific and Technical Reports no. EUR 24816 EN – 2011. Institute for Energy, Joint Research Centre, European Commission. Available online at http://publications.jrc.ec.europa.eu/repository/bitstream/111111111/22908/1/reqno_jrc64429_cirtical%20issues%20in%20estimating%20iluc%20emissions%20pdf%20version.pdf

Marshall, E., M. Caswell, S. Malcolm, M. Motamed, J. Hrubovcak, C. Jones, and C. Nickerson (2011). "Measuring the Indirect Land-Use Change Associated With Increased Biofuel Feedstock Production: A Review of Modeling Efforts—Report to Congress." Administrative Publication no. AP-054. Economic Research Service, U.S. Department of Agriculture. February. Available online at http://www.ers.usda.gov/publications/ap-administrative-publication/ap-054.aspx

Meyers, W. H., P. Westhoff, J. F. Fabiosa, and D. J. Hayes (2010). "The FAPRI Global Modeling System and Outlook Process." *Journal of International Agricultural Trade and Development* 6, no. 1 : 1–19.

Miettinen, J., A. Hooijer, D. Tollenaar, S. Page, C. Malins, R. Vernimmen, C. Shi, and S.C. Liew (2012). *Historical Analysis and Projection of Oil Palm Plantation Expansion on Peatland in Southeast Asia.* White Paper no. 17. Indirect Effects of Biofuel Production Series. International Council on Clean Transportation. February. Available online at http://www.theicct.org/sites/default/files/publications/ICCT_palm-expansion_Feb2012.pdf

Mills, J. A. N., J. Dijkstra, A. Bannink, S. B. Cammell, E. Kebreab, and J. France (2001). "A Mechanistic Model of Whole-Tract Digestion and Methanogenesis in the Lactating Dairy Cow: Model Development, Evaluation, and Application." *Journal of Animal Science* 79, no. 6 (June): 1584–97.

Mitchell, D. (2008). "A Note on Rising Food Prices." Policy Research Working Paper no. 4682. Development Prospects Group, World Bank. July. Available online at http://www-wds.worldbank.org/servlet/WDSContentServer/WDSP/IB/2008/07/28/000020439_20080728103002/Rendered/PDF/WP4682.pdf

Moe, P. W., and H. F. Tyrrell (1979). "Methane Production in Dairy Cows." *Journal of Dairy Science* 62, no. 10 (October): 1583–86.

Morton, D. C., R. S. DeFries, J. T. Randerson, L. Giglio, W. Schroeder, and G. R. van der Werf (2008). "Agricultural Intensification Increases Deforestation Fire Activity in Amazonia." *Global Change Biology* 14, no. 10 (October): 2262–75.

Morton, D. C., R. S. DeFries, Y. E. Shimabukuro, L. O. Anderson, E. Arai, F. del Bon Espirito-Santo, R. Freitas, and J. Morisette (2006). "Cropland Expansion Changes Deforestation Dynamics in the Southern Brazilian Amazon." *Proceedings of the National Academy of Sciences* 103, no. 39 (September 26): 14637–41.

Mosier, A. R., J. M. Duxbury, J. R. Freney, O. Heinemeyer, and K. Minami (1996). "Nitrous Oxide Emissions from Agricultural Fields: Assessment, Measurement and Mitigation. *Plant and Soil* 181, no. 1 (April): 95–108.

Murdiyarso, D., K. Hergoualc'h, and L. V. Verchot (2010). "Opportunities for Reducing Greenhouse Gas Emissions in Tropical Peatlands." *Proceedings of the National Academy of Science* 107, no. 46 (November 16): 19655–60.

Murty, D., M. U. F. Kirschbaum, R. E. McMurtrie, and H. McGilvray (2002). "Does Conversion of Forest to Agricultural Land Change Soil Carbon and Nitrogen? A Review of the Literature." *Global Change Biology* 8, no. 2 (February): 105–23.

Myneni, R. B., J. Dong, C. J. Tucker, R. K. Kaufmann, P. E. Kauppi, J. Liski, L. Zhou, V. Alexeyev, and M. K. Hughes (2001). "A Large Carbon Sink in the Woody Biomass of Northern Forests." *Proceedings of the National Academy of Science* 98, no. 26 (December 18): 14784–89.

Nerlove, M. (1956). "Estimates of the Elasticities of Supply of Selected Agricultural Commodities." *Journal of Farm Economics* 38, no. 2 (May): 496–509.

Norton, N. A. (1986). "The Effect of Acreage Reduction Programs on the Production of Corn, Wheat and Cotton: A Profit Function Approach." Master's thesis, University of Maryland.

Novoa, R. S. A., and H. R. Tejeda (2006). "Evaluation of the N_2O Emissions from N in Plant Residues as Affected by Environmental and Management Factors." *Nutrient Cycling in Agroecosystems* 75, nos. 1–3 (July): 29–46.

O'Hara, G. W., and R. M. Daniel (1985). "Rhizobial Denitrification: A Review." *Soil Biology and Biochemistry* 17, no. 1: 1–9.

Organisation for Economic Co-operation and Development (OECD) (2001). *Market Effects of Crop Support Measures.* Paris: OECD.

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

Organisation for Economic Co-operation and Development and Food and Agriculture Organization of the United Nations (OECD-FAO) (2012). OECD-FAO Agricultural Outlook 2012–2021. 2012 edition, retrieved November 29, 2012. Most recent edition available online at http://www.oecd.org/site/oecd-faoagriculturaloutlook/

Ogg, C. W., S.-E. Webb, and W.-Y. Huang (1984). "Cropland Acreage Reduction Alternatives: An Economic Analysis of a Soil Conservation Reserve and Competitive Bids." *Journal of Soil and Water Conservation* 39, no. 6 (November/December): 379–83.

Oladosu, G., K. Kline, R. Uria-Martinez, and L. Eaton (2011). "Sources of Corn for Ethanol Production in the United States: A Decomposition Analysis of the Empirical Data." *Biofuels, Bioproducts and Biorefining* 5, no. 6 (November/December): 640–53.

Oregon Department of Environmental Quality (2011). *Oregon Low Carbon Fuel Standards: Advisory Committee Process and Program Design.* Final Report. January 25. Available online at http://www.deq.state.or.us/aq/committees/docs/lcfs/reportFinal.pdf

Overmars, K. P., E. Stehfest, J. P. M. Ros, and A. G. Prins (2011). "Indirect Land Use Change Emissions Related to EU Biofuel Consumption: An Analysis Based on Historical Data." *Environmental Science and Policy* 14, no. 3 (May): 248–57.

Page, S. E., R. Morrison, C. Malins, A. Hooijer, J. O. Rieley, and J. Jauhiainen (2011). *Review of Peat Surface Greenhouse Gas Emissions from Oil Palm Plantations in Southeast Asia.* White Paper no. 15. Indirect Effects of Biofuel Production Series.International Council on Clean Transportation. September. Available online at http://www.theicct.org/sites/default/files/publications/ICCT_Peat-Emissions_Sept2011.pdf

Page, S. E., J. O. Rieley, and C. J. Banks (2011). "Global and Regional Importance of the Tropical Peatland Carbon Pool." *Global Change Biology* 17, no. 2 (February): 798–818.

Pan, Y., R. A. Birdsey, J. Fang, R. Houghton, P. E. Kauppi, W. A. Kurz, O. L. Phillips, et al. (2011). "A Large and Persistent Carbon Sink in the World's Forests." *Science* 333, no. 6045 (August 19): 988–93.

Parikh, J. K., and G. Krömer (1985). "Modeling Energy and Agriculture Interactions—II: Food-Fodder-Fuel-Fertilizer Relationships for Biomass in Bangladesh." *Energy* 10, no. 7 (July): 805–17.

Parkin, T. B., and T. C. Kaspar (2006). "Nitrous Oxide Emissions from Corn-Soybean Systems in the Midwest." *Journal of Environmental Quality* 35, no. 4 (July–August): 1496–1506.

Parry, M. L., C. Rosenzweig, A. Iglesias, M. Livermore, and G. Fischer (2004). "Effects of Climate Change on Global Food Production under SRES Emissions and Socio-Economic Scenarios." *Global Environmental Change* 14, no. 1 (April): 53–67.

Plevin, R. J., M. O'Hare, A. D. Jones, M. S. Torn, and H. K. Gibbs (2010). "Greenhouse Gas Emissions from Biofuels' Indirect Land Use Change Are Uncertain but May Be Much Greater than Previously Estimated." *Environmental Science and Technology* 44, no. 21 (November 1): 8015–21.

Porter, J. R., and M. A. Semenov (2011). "Crop Responses to Climatic Variation." *Philosophical Transactions of the Royal Society—Series B: Biological Sciences* 360, no. 1463 (November 29): 2021–35.

Prévosto, B., L. Kuiters, M. Bernhardt-Römermann, M. Dölle, W. Schmidt, M. Hoffmann, J. Van Uytvanck, et al. (2011). "Impacts of Land Abandonment on Vegetation: Successional Pathways in European Habitats." *Folia Geobotanica* 46, no. 4 (December): 303–25.

Proctor, J., J. M. Anderson, S. C. L. Fogden, and H. W. Vallack (1983). "Ecological Studies in Four Contrasting Lowland Rain Forests in Gunung Mulu National Park, Sarawak: II. Litterfall, Litter Standing Crop and Preliminary Observations on Herbivory." *Journal of Ecology* 71, no. 1 (March): 261–83.

Rahmstorf, S., and D. Coumou (2011). "Increase of Extreme Events in a Warming World." *Proceedings of the National Academy of Sciences* 108, no. 44 (November 1): 17905–9.

Rajagopal, D., G. Hochman, and D. Zilberman (2011). "Indirect Fuel Use Change (IFUC) and the Lifecycle Environmental Impact of Biofuel Policies." *Energy Policy* 39, no. 1 (January): 228–33.

Reay, D. S., E. A. Davidson, K. A. Smith, P. Smith, J. M. Melillo, F. Dentener, and P. J. Crutzen (2012). "Global Agriculture and Nitrous Oxide Emissions." *Nature Climate Change* 2, no. 6 (June): 410–16.

Reich, P. B. (2011). "Biogeochemistry: Taking Stock of Forest Carbon." *Nature Climate Change* 1, no. 10 (October): 346–47.

Reilly, J. M. (2010). "GTAP-BIO-ADV and Land Use Emissions from Expanded Biofuels Production." Paper written for the Air Resources Board (California). November 4. Available online at http://www.arb.ca.gov/fuels/lcfs/workgroups/ewg/010511-reilly-rpt.pdf

Repetto, R., and D. Austin (1997). *The Costs of Climate Protection: A Guide for the Perplexed.* Climate Protection Initiative. World Resources Institute. Available online at http://www.wri.org/sites/default/files/pdf/costsclimateprotection_bw.pdf

Roberson, K.D (2003). Use of dried distillers' grains with solubles in growing-finishing diets of turkey hens. International Journal of Poultry Science, 2 (6), 389-393.

Roberts, M. J., and W. Schlenker (2010). "Identifying Supply and Demand Elasticities of Agricultural Commodities: Implications for the US Ethanol Mandate." Working Paper no. 15921. NBER Working Paper Series. National Bureau of Economic Research. April. Available online at http://www.nber.org/papers/w15921.pdf?new_window=1

Rochette, P., D. A. Angers, G. Belanger, M. H. Chantigny, D. Prévost, and G. Lévesque (2004). "Emissions of N_2O from Alfalfa and Soybean Crops in Eastern Canada." *Soil Science Society of America Journal* 68, no. 2 (March–April): 493–506.

Rosas, F. (2011). "World Fertilizer Model—The WorldNPK Model." Working Paper no. 11-WP 520. Center for Agricultural and Rural Development, Iowa State University. April. Available online at http://ageconsearch.umn.edu/bitstream/103223/2/11-WP_520.NEW.pdf

Rosegrant, M. W., C. Ringler, S. Msangi, T. B. Sulser, T. Zhu, and S. A. Cline (2008). "International Model for Policy Analysis of Agricultural Commodities and Trade (IMPACT): Model Description." International Food Policy Research Institute. June. Available online at http://www.ifpri.org/sites/default/files/publications/impactwater.pdf

Saggar, S., N. S. Bolan, R. Bhandral, C. B. Hedley, and J. Luo (2004). "A Review of Emissions of Methane, Ammonia, and Nitrous Oxide from Animal Excreta Deposition and Farm Effluent Application in Grazed Pastures." *New Zealand Journal of Agricultural Research* 47, no. 4: 513–44.

Sanchez, S. T., J. Woods, M. Akhurst, M. Brander, M. O'Hare, T. P. Dawson, R. Edwards, A. J. Liska, and R. Malpas (2012). "Accounting for Indirect Land-Use Change in the Life Cycle Assessment of Biofuel Supply Chains." *Journal of the Royal Society Interface* 9, no. 71 (June 7): 1105–19.

Schumacher, E. F. (1977). *A Guide for the Perplexed.* New York: Harper Perennial.

Seale Jr., J., A. Regmi, and J. Bernstein (2003). "International Evidence on Food Consumption Patterns." Technical Bulletin no. 1904. Economic Research Service, U.S. Department of Agriculture.: October. Available online at http://www.ers.usda.gov/publications/tb-technical-bulletin/tb1904.aspx#.UhOjDGRgZ74

Searchinger, T., R. Heimlich, R. A. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, D. Hayes, and T.-H. Yu (2008). "Use of U.S. Croplands for Biofuels Increases Greenhouse Gases through Emissions from Land-Use Change." *Science* 319, no. 5867 (February 29): 1238–40.

Searle, S., and C. Malins (2011). "Estimates of Carbon Storage in Wood Products Following Land Clearing." Working paper no. 2011-4. International Council on Clean Transportation. August. Available online at http://www.theicct.org/sites/default/files/publications/ICCT_carbon_storage_in_wood_products_August_2011.pdf

Shurson, J. (2009a). "Analysis of current feeding practices of distiller's grains with solubles in livestock and poultry feed relative to land use credits associated with determining the Low Carbon Fuels Standard for ethanol." Renewable Fuel Association. Available in http://www.ethanolrfa.org/page//objects/documents/2290/shurson.pdf?nocdn=1

Shurson, J. (2009b). "Impact of Distillers' Grains on Indirect Land Use." Presentation at the National Corn Grower Land Use Conference, St. Louis, MO, August 25- 26, 2009. Retrieved from http://www.ddgs.umn.edu/pptswine/2009Shurson%20Impact%20of%20distillers%20grains%20on%20indirect%20land%20use%20change.pdf

Shurson, J., S. Mindy, W. Jennifer, and W. Mark (2003). "Value and Use of 'new generation' distillers' dried grains with solubles in swine diets." Presented at the 19th International Alltech Conference, Lexington, KY, May 13, 2003. Retrieved from http://www.ddgs.umn.edu/prod/groups/cfans/@pub/@cfans/@ddgs/documents/asset/cfans_asset_413196.pdf

Shurson, J., and S. Mindy (2002). "Feeding recommendations and example diets containing Minnesota-South Dakota produced DDGS for Swine." Minnesota: Department of Animal Science, University of Minnesota. Retrieved from http://www.ddgs.umn.edu/prod/groups/cfans/@pub/@cfans/@ddgs/documents/asset/cfans_asset_411731.pdf

Slayton, T. (2009). "Rice Crisis Forensics: How Asian Governments Carelessly Set the World Rice Market on Fire." Working Paper no. 163. Center for Global Development. March. Available online at http://ssrn.com/abstract=1392418

Sneath, R. W., V. R. Phillips, T. G. M. Demmers, L. R. Burgess, J. L. Short, and S. K. Welch (1997). "Long-term Measurements of Greenhouse Gas Emissions from UK Livestock Buildings." In *Livestock Environment V. Vol. I. Proceedings of the Fifth International Symposium.* (Bloomington, Minnesota, May 29–31). Ed. R. W. Bottcher and S. J. Hoff. St. Joseph, Michigan: American Society of Agricultural Engineers. Pp. 146–53.

Sohngen, B., and R. Mendelsohn (2007). "A Sensitivity Analysis of Forest Carbon Sequestration." In *Human-Induced Climate Change: An Interdisciplinary Assessment.* Ed. M. E. Schlesinger, H. S. Kheshgi, J. Smith, F. C. de la Chesnaye, J. M. Reilly, T. Wilson, and C. Kolstad. Cambridge: Cambridge University Press. Pp. 227–37.

Stott, D. E., D. R. Smith, and D. L. Bucholtz (2007). "GHG Emissions from Corn-Soybean Rotations on a Mollisol." Poster presented at the 4th USDA Greenhouse Gas Symposium. Baltimore. February 5–8. Available online at http://soilcarboncenter.k-state.edu/conference_2007/pdf_files_2007/Diane_Stott_2.pdf

Taheripour, F., T. W. Hertel, W. E. Tyner, J. F. Beckman, and D. K. Birur (2008). "Biofuels and Their By-Products: Global Economic and Environmental Implications." Department of Agricultural Economics, Purdue University. June. Available online at https://www.gtap.agecon.purdue.edu/resources/download/3974.pdf

Taheripour, F., W. E. Tyner, and M. Q. Wang (2011). “Global Land Use Changes Due to the U.S. Cellulosic Biofuel Program Simulated with the GTAP Model.” Department of Agricultural Economics, Purdue University, and Argonne National Laboratory, U.S. Department of Energy.

Takahashi, M., S. Ishizuka, S. Ugawa, Y. Sakai, H. Sakai, K. Ono, S. Hashimoto, Y. Matsuura, and K. Morisada (2010). “Carbon Stock in Litter, Deadwood and Soil in Japan’s Forest Sector and its Comparison with Carbon Stock in Agricultural Soils.” *Soil Science and Plant Nutrition* 56, no. 1 (February): 19–30.

Timmer, C. P. (2009). “Did Speculation Affect World Rice Prices?” ESA Working Paper no. 09-07. Agricultural Development Economics Division, Food and Agriculture Organization of the United Nations. April. Paper originally prepared for the FAO conference on “Rice Policies in Asia.” Chiang Mai, Thailand. February 9–12. Available online at ftp://ftp.fao.org/docrep/fao/011/ak232e/ak232e00.pdf

Tipper, R., Hutchison, C. and Brander, M. (2009). “A Practical Approach for Policies to Address GHG Emissions from Indirect Land Use Change Associated with Biofuels”, Ecometrica.

Tokgoz, S., A. Elobeid, J. F. Fabiosa, D. J. Hayes, B. A. Babcock, T.-H. Yu, F. Dong, C. E. Hart, and J. C. Beghin (2007). *Emerging Biofuels: Outlook of Effects on U.S. Grain, Oilseed, and Livestock Markets.* Staff Report no. 07-SR 101. Revised version. Center for Agricultural and Rural Development, Iowa State University. July. Available online at http://www.card.iastate.edu/publications/dbs/pdffiles/07sr101.pdf

Toma, Y., and R. Hatano (2007). “Effect of Crop Residue C:N Ratio on N_2O Emissions from Gray Lowland Soil in Mikasa, Hokkaido, Japan.” *Soil Science and Plant Nutrition* 53, no. 2 (April): 198–205.

Trenkle, A. (2003). “Evaluation of wet and dry distillers grains with solubles for finishing Holstein steers.” Report to the Iowa Corn Promotion Board. Retrieved from http://www.ddgs.umn.edu/articles-beef/2003-Trenkle-%20FINAL%20ISU12-03.pdf

Turner, D. P., G. J. Koerper, M. E. Harmon, and J. J. Lee (1995). A carbon budget for forests of the conterminous United States. Ecological Applications, 5(2), 421-436.

Tyner, W. E., F. Taheripour, Q. Zhuang, D. Birur, and U. Baldos (2010). *Land Use Changes and Consequent CO_2 Emissions Due to US Corn Ethanol Production: A Comprehensive Analysis.* Final Report. Department of Agricultural Economics, Purdue University. April. Available online at https://ethanol.org/pdf/contentmgmt/Purdue_new_ILUC_report_April_2010.pdf

UK Department for Environment, Food and Rural Affairs (UK DEFRA) (2001). “Economic Evaluation of Set-Aside.” Report to DEFRA by the Centre for Rural Economic Research, University of Cambridge. Available online at http://archive.defra.gov.uk/evidence/economics/foodfarm/evaluation/setaside/fullrep.pdf

UK Renewable Fuels Agency (UK RFA) (2011). “Verified RFA Quarterly Report 8: 15 April 2009–14 April 2010.” Available online at https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/69729/Verified_data_2009-10.pdf

UK Renewable Fuels Agency (UK RFA) (2008). *The Gallagher Review of the Indirect Effects of Biofuels Production.* July. Available online at http://www.unido.org/fileadmin/user_media/UNIDO_Header_Site/Subsites/Green_Industry_Asia_Conference__Maanila_/GC13/Gallagher_Report.pdf

United Nations, Office of the UN High Commissioner for Human Rights, Mandate of the Special Rapporteur on the Right to Food (2011). “Observations on the Current Food Price Situation: Background Note.” January 21. Available online at http://www.srfood.org/images/stories/pdf/otherdocuments/20110124_background-note-food-price-situation_en.pdf

U.S. Congress. House. Agricuture Committee (2009). Statement of R. Brooke Coleman, executive director, New Fuels Alliance, before the Subcommittee on Conservation, Credit, Energy and Research. 111th Cong., 1st sess. May 6. Available online at https://agriculture.house.gov/sites/republicans.agriculture.house.gov/files/testimony/110/h90506/Coleman.pdf

U.S. Environmental Protection Agency (U.S. EPA). (2009). Regulation of Fuels and Fuel Additives: Changes to Renewable Fuel Standard Program; Proposed Rule (p 24904-25143). Federal Register: EPA-HQ-OAR-2005-0161; FRL-8903-1.

U.S. Environmental Protection Agency (U.S. EPA), Office of Transportation and Air Quality, Assessment and Standards Division (2010a). *Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis.* EPA-420-R-10-006. February. Available online at http://www.epa.gov/otaq/renewablefuels/420r10006.pdf

U.S. Environmental Protection Agency (U.S. EPA). (2010b). Regulation of Fuels and Fuel Additives: Changes to Renewable Fuel Standard Program (p 14670-14904). Federal Register: EPA-HQ-OAR-2005-0161; FRL-9112-3.

U.S. Environmental Protection Agency (U.S. EPA). (2010c). Supplemental Determination for Renewable Fuels Produced Under the Final RFS2 Program From Canola Oil (p 59622-59634). Federal Register: EPA-HQ-OAR-2005-0161; FRL-9207-1.

U.S. Environmental Protection Agency (U.S. EPA). (2012). Notice of Data Availability Concerning Renewable Fuels Produced From Palm Oil Under the RFS Program (p 4300-4318). Federal Register: EPA-HQ-OAR-2005-0161; FRL-9608-8.

U.S. Environmental Protection Agency, Office of the Administrator, Science Advisory Board (U.S. EPA SAB) (2012). *SAB Review of EPA’s Accounting Framework for Biogenic CO_2 Emissions from Stationary Sources.* EPA-SAB-12-011. September. Available online at http://yosemite.epa.gov/sab/sabproduct.nsf/0/57B7A4F1987D7F7385257A87007977F6/$File/EPA-SAB-12-011-unsigned.pdf

Vander Pol, K. J., G. Erickson, M.A. Greenquist, T.J. Klopfenstein, and T. Robb (2006). “Effect of dietary inclusion of wet distillers grains on feedlot performance of finishing cattle and energy value relative to corn. Nebraska Beef Cattle Reports, Lincoln: Animal Science Department, University of Nebraska-Lincoln.

Wagner-Riddle, C., A. Furon, N. L. McLaughlin, I. Lee, J. Barbeau, S. Jayasundara, G. Parkin, P. Von Bertoldi, and J. Warland (2007). “Intensive Measurement of Nitrous Oxide Emissions from a Corn-Soybean-Wheat Rotation under Two Contrasting Management Systems over 5 Years.” *Global Change Biology* 13, no. 8 (August): 1722–36.

Wagner-Riddle, C., G. W. Thurtell, G. K. Kidd, E. G. Beauchamp, and R. Sweetman (1997). “Estimates of Nitrous Oxide Emissions from Agricultural Fields over 28 Months.” *Canadian Journal of Soil Science* 77, no. 2 (May): 135–44.

Wang, L., and J. Davis (1998). “Can China Feed Its People into the Next Millennium? Projections for China’s Grain Supply and Demand to 2010.” *International Review of Applied Economics* 12, no. 1: 53–67.

Wang, M. Q., J. Han, Z. Haq, W. E. Tyner, M. Wu, and A. Elgowainy (2011). "Energy and Greenhouse Gas Emission Effects of Corn and Cellulosic Ethanol with Technology Improvements and Land Use Changes." *Biomass and Bioenergy* 35, no. 5 (May): 1885–96.

Wang, M., J. Han, J. B. Dunn, H. Cai, and A. Elgowainy (2012). "Well-to-Wheels Energy Use and Greenhouse Gas Emissions of Ethanol from Corn, Sugarcane and Cellulosic Biomass for US Use." *Environmental Research Letters* 7, no. 4 (December): 045905.

Weisgerber, P. (1969). "Productivity of Diverted Crop Land." Report no. ERS-398. Economic Research Service, U.S. Department of Agriculture. April.

Wergen, G., and J. Krug (2010). "Record-Breaking Temperatures Reveal a Warming Climate." *Europhysics Letters* 92, no. 3 (November): 30008.

Wiggins, S., S. Keats, and J. Compton (2010). "What caused the food price spike of 2007/08? Lessons for world cereals markets." Food Prices Project Report, Overseas Development Insitute, London, UK.

Wiggins, S., and S. Levy (2008). "Rising Food Prices: A Global Crisis." ODI Briefing Paper no. 37. Overseas Development Institute. April. Available online at http://www.odi.org.uk/sites/odi.org.uk/files/odi-assets/publications-opinion-files/1630.pdf

Wise, T. A., and S. Murphy (2012). "Resolving the Food Crisis: Assessing Global Policy Reforms since 2007." Institute for Agriculture and Trade Policy and the Global Development and Environment Institute at Tufts University. January. Available online at http://www.iatp.org/files/2012_01_17_ResolvingFoodCrisis_SM_TW.pdf

Witzke, P., J. Fabiosa, S. H. Gay, A. Golub, P. Havlik, S. Msangi, S. Tokgoz, and T. Searchinger (2010). "A Decomposition Approach to Assess ILUC Results from Global Modeling Efforts." Paper presented at the International Agricultural Trade Research Consortium (IATRC) Symposium. Stuttgart. June 26–29. Available online at https://www.uni-hohenheim.de/fileadmin/einrichtungen/iatrc2010/Papers/Witzke_et_al._IATRC_Summer_2010.pdf

Wolfram MathWorld (n.d.). Available online at http://mathworld.wolfram.com/

World Bank (2008). "World Bank Group President, Robert Zoellick, Opening Press Briefing at the WB-IMF Spring Meetings 2008.". April 10. Available online at http://web.worldbank.org/WBSITE/EXTERNAL/NEWS/0,,contentMDK:21725341~pagePK:64257043~piPK:437376~theSitePK:4607,00.html

World Wildlife Fund (WWF) (2012). "Terrestrial Ecoregions." Available online at http://worldwildlife.org/biome-categories/terrestrial-ecoregions

Wright, P. G. (1928). *The Tariff on Animal and Vegetable Oils.* New York: Macmillan.

Yang, L., and Z. Cai (2005). "The Effect of Growing Soybean (*Glycine max. L.*) on N_2O Emission from Soil." *Soil Biology and Biochemistry* 37, no. 6 (June): 1205–9.

Yang, Y., J. Bae, J. Kim, and S. Suh (2012). "Replacing Gasoline with Corn Ethanol Results in Significant Environmental Problem-Shifting." *Environmental Science and Technology* 46, no. 7 (April 3): 3671–78.

Yeh, S., S. M. Jordaan, A. R. Brandt, M. R. Turetsky, S. Spatari, and D. W. Keith (2010). "Land use greenhouse gas emissions from conventional oil production and oil sands." Environmental science & technology, 44(22), 8766-8772.

Yevich, R., and J. A. Logan (2003). "An Assessment of Biofuel Use and Burning of Agricultural Waste in the Developing World." *Global Biogeochemical Cycles* 17, no. 4 (December): 1095.

Zilberman, D., G. Hochman, and D. Rajagopal (2010). "Indirect Land Use Change: A Second-best Solution to a First-class Problem." *AgBioForum: The Journal of Agrobiotechnology Management and Economics* 13, no. 4: 382–90.

APPENDIX A. THE REVISED WITZKE DECOMPOSITION

The basic decomposition proposed by Witzke et al. (2010) is

$$\Delta l_{ri} = \frac{\Delta n_{ri}}{y_{ri}^{0}} + \frac{\Delta d_{ri}^{T}}{y_{ri}^{0}} - \frac{l_{ri}^{1}\Delta y_{ri}}{y_{ri}^{0}} \tag{1}$$

where l is land use, n is change in net trade, and y is yield, with 0 and 1 indexing baseline and increased biofuels scenario respectively, while i and r index commodities and regions respectively. Demand, d^{T} , includes both biofuels and other alimentary needs (food and fodder), so one can split it further into $d_{ri}^{T} = B_{ri} + d_{ri}$, where B_{ri} is biofuel feedstock demand and d_{ir} is other demand, to give

$$\Delta l_{ri} = \frac{\Delta B_{ri}}{Y_{i}^{0}} + \frac{\Delta n_{ri}}{y_{ri}^{0}} + \frac{\Delta d_{ri}}{y_{ri}^{0}} - \frac{l_{ri}^{1}\Delta y_{ri}}{y_{ri}^{0}} \tag{2}$$

By introducing a conversion matrix a_{ri} (defined only for the initial, 0 , yield state), one can transform local yield into average global yield for a commodity through the relation

$$a_{ri} = \frac{y_{ri}^{0}}{y_{i}^{0}} \tag{3}$$

in which y_{ri}^{0} is the global average yield for commodity i . a_{ri} is the ratio of local to global average yield, that is, $a_{ri} > 1$ is a below-average yield, and $a_{ri} > 1$ is an above-average yield. Multiplying both sides of Equation 2 by a_{ri} gives

$$a_{ir}\,\Delta l_{ir} = \frac{\Delta B_{ir}}{Y_{i}^{0}} + \frac{\Delta d_{ir}}{Y_{i}^{0}} + \frac{\Delta n_{ir}}{Y_{i}^{0}} - \frac{l_{ir}^{1}\Delta y_{ir}}{Y_{i}^{0}} \tag{4}$$

for which the left-hand side can be renamed in this way

$$a_{ri}\,\Delta l_{ri} = \Delta(a_{ri}l_{ri}) = a_{ri}l_{ri}^{1} - a_{ri}l_{ri}^{0} = \Delta L_{ri} \tag{5}$$

where Δl_{ir} is the change in the land use required at initial global average yield. The right-hand side of Equation 4 is now a decomposition between increased biofuel use, change in non-biofuel-related feedstock consumption, net trade, and change related to yield variation, becoming

$$\Delta L_{ri} = \frac{\Delta B_{ri}}{Y_{i}^{0}} + \frac{\Delta d_{ri}}{Y_{i}^{0}} + \frac{\Delta n_{ri}}{Y_{i}^{0}} - \frac{l_{ri}^{1}\Delta y_{ri}}{Y_{i}^{0}}\ . \tag{6}$$

In general, it should be possible to identify the tonnage of by-products produced because of increased biofuel production in each model, where Δbp_{ri} is the increase in the substitution of commodity i in region r by biofuel by-products. Either the model will explicitly produce this number, or it can be calculated/estimated as

$$\Sigma_{ri}\Delta bp_{ri} = \Sigma_{rf}\beta_{rf}\Delta B_{rf} \tag{7}$$

where β is the ratio of metric tons of by-product to tons of biofuel produced, and f indexes the feedstock from which the biofuel is produced (since the feedstock replaced is not necessarily the same as the one demanded for production). Where scenarios deal with biofuel from only one feedstock, and if by-product output ratios are assumed to be geographically constant, β will be a single number rather than a tensor.

It is mathematically possible to equate the sums in equation 7 but potentially more challenging to determine exactly which feedstocks are being replaced by by-products. Providing, however, that it is possible to assign either exact or approximate substitution ratios ϕ of the fraction of by-product from a biofuel feedstock f that will replace a commodity i in a region r,

$$\phi_{ri}^{f} = \frac{\Delta bp_{ri}^{f}}{\beta_{rf}\Delta B_{rf}} \tag{8}$$

it will be possible to allocate exactly or approximately by-product replacement to specific commodities

$$\Delta bp_{ri} = \Sigma\phi_{ri}^{f}\beta_{rf}\Delta B_{rf}\,. \tag{9}$$

Once a value has been set for bp_{ri}, domestic demand change can be further broken down into demand change due to availability of by-products as replacement commodities and demand change due to reduced food consumption ('real' demand change, Δc_{ri}) $\Delta d_{ri} = \Delta c_{ri} - \Delta bp_{ri}$. This gives

$$\Delta L_{ri} = \frac{\Delta B_{ri}}{Y_i^{\,0}} + \frac{\Delta c_{ri}}{Y_i^{\,0}} - \frac{\Delta bp_{ri}}{Y_i^{\,0}} + \frac{\Delta n_{ri}}{Y_i^{\,0}} - \frac{l_{ri}^{1}\Delta y_{ri}}{Y_i^{\,0}} \tag{10}$$

Part of the change in land demand can be understood as being a result of regional productivity variation (which can also be referred to as area switching)—the extensification/intensification resulting from locating additional production in region with a higher or lower than global average yield. One approach to evaluating the amount of land saved or required because of crop switching would be defining a change in production attributable to area switching, S

$$S_{ri} = \Delta l_{ri} - \Delta L_{ri} = \Delta l_{ri}\left(1 - \frac{Y_{ri}^{\,0}}{Y_i^{\,0}}\right) \tag{11}$$

One can also express S more lengthily as a function of the land requirements for the other elements at either local or global average yields

$$S_{ri} = \left(\frac{\Delta B_{ri} + \Delta c_{ri} - \Delta bp_{ri} + \Delta n_{ri} + l_{ri}^{1}\Delta y_{ri}}{y_{ri}^{\,0}} - \frac{\Delta B_{ri} + \Delta c_{ri} - \Delta bp_{ri} + \Delta n_{ri} + l_{ri}^{1}\Delta y_{ri}}{Y_i^{\,0}}\right). \tag{12}$$

Including this term gives a full regional decomposition for the actual area change within a specified region

$$\Delta l_{ri} = \frac{\Delta B_{ri}}{Y_i^{\,0}} + \frac{\Delta c_{ri}}{Y_i^{\,0}} - \frac{\Delta bp_{ri}}{Y_i^{\,0}} + \frac{\Delta n_{ri}}{Y_i^{\,0}} - \frac{l^1_{ri}\Delta y_{ri}}{Y_i^{\,0}} + S_{ri}\,. \tag{13}$$

Now consider the global outcomes of the decomposition. For the sake of simplifying the notation, for each variable v in the set of variables $\{B, c, bp, n\}$, subsequent equations shall denote $\Sigma_{ri}\frac{\Delta v_{ri}}{Y_{ri}}$ as Δv , $\Sigma_{ri}\frac{l^1_{ri}\Delta y_{ri}}{Y_i}$ as Δy , $\Sigma_{ri}S_{ri}$ as S, $\Sigma_{ri}\Delta L_{ri}$ as ΔL , and $\Sigma_{ri}\Delta l_{ri}$ as Δl . This then gives, across all commodities and regions at world average yield, an overall decomposition

$$\Delta L = \Delta B + \Delta c - \Delta bp + \Delta n - l^1\Delta y, \tag{14}$$

which can be transformed to actual overall land use change as

$$\Delta l = \Delta B + \Delta c - \Delta bp - l^1\Delta y + S. \tag{15}$$

In this final formulation, Δl is the total land use change, ΔB is the land required to meet biofuel demand, Δc is the land conserved by changes in global consumption of food, fodder, etc., Δbp is the reduction in land requirement from using by-products, Δn goes to zero by hypothesis (as, globally, net trade must be zero), $l^1\Delta y$ is the change in land requirement attributable to demand-induced intensification/extensification within each region, and S represents the land required or saved because additional production takes place at lower or higher than global average yield.

APPENDIX B. DOCUMENTATION FOR EMISSION FACTORS APPLIED TO THE WITZKE DECOMPOSITION

B.1. SOIL CARBON

Soil carbon stocks were taken from the International Institute for Applied Systems Analysis's Harmonized World Soil Database (http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/HWSD_Data.html?sb=4), which provides soil carbon concentrations in the top 30 cm layer for more than 16,000 soil mapping units globally. Biomes from the World Wildlife Fund (WWF) Terrestrial Ecoregions Map (http://www.worldwildlife.org/science/data/item6373.html) were then mapped onto the world soil map. Average soil carbon stocks for each biome within each country were calculated. The Witzke decomposition regions were combined with the Searchinger regions (Searchinger et al. 2008) to create subregions. For instance, within Europe, the Witzke regions are 'EU27' and 'Other Europe,' while the Searchinger regions are 'Europe' and 'Former Soviet Union;' the subregions are then 'EU27-Europe,' 'Other Europe–Europe,' and 'Other Europe–Former Soviet Union.' Soil carbon stocks within each biome were aggregated for these subregions by taking the weighted average by land area. The land extension coefficients given in Searchinger et al. were then applied to each biome and subregion. The soil carbon stocks were multiplied by the land extension coefficients and then summed within each subregion for a total soil carbon emission factor per subregion. Subregion soil carbon stocks were then aggregated into Witzke regions by taking the weighted average by land area. Regional emission factors were created by multiplying regional soil carbon stocks by 0.25. The worldwide emission factor was created by taking the weighted average of all regional soil carbon stocks by regional area. The 'Rest of World' emission factor was created by taking the weighted average of all regional soil carbon stocks, excluding the United States and the EU27, by regional area.

B.2. VEGETATION CARBON

B.2.1. Vegetation emissions

Data were taken from the Intergovernmental Panel on Climate Change 2006 Tier 1 default values (IPCC 2006a; 2006b; 2006c); these included aboveground live biomass (metric tons of dry matter per hectare), aboveground forest growth (tons of dry matter per hectare per year), root-to-shoot ratio (tons of dry matter per hectare), and dead litter (tons of carbon per hectare), each parameter given by climate and ecological system. Aboveground dry biomass stocks were multiplied by (1 + root-to-shoot ratio) to account for belowground vegetation biomass. Total live biomass per hectare was multiplied by 0.47 to convert to carbon per hectare. Dead litter carbon (C) stocks were added to live vegetation C stocks for total vegetation C per hectare. Total C for forest systems was multiplied by 0.9 for developed regions (United States, Europe, Pacific Developed) and by 0.96 for developing regions (all other regions) to calculate C emissions, accounting for harvested wood products (Searle and Malins 2011). The resulting total C emissions per climate/ecological system were assigned to subregions (as described above for soil C). The land extension coefficients from Searchinger et al. (2008) were then applied to each climate/ecological system for each subregion. Total C emissions were summed across each climate/ecological system for each subregion. Subregions were then aggregated to Witzke regions as described above for soil carbon. Worldwide and Rest of World emissions factors were calculated as above for soil carbon.

B.2.2. Vegetation sequestration

Vegetation growth was calculated by multiplying the aboveground forest growth values from the IPCC Tier 1 default values by (1 + root-to-shoot ratio) to account for below-ground vegetation growth. The same land extension coefficients used for emissions were also used for sequestration; thus, it is assumed that forest regrowth (conversion of cropland to forestland) is most likely to occur in areas experiencing forest-to-cropland conversion. This is a simplified assumption but is relatively reasonable considering that all cropland, whether new or reverting to forest, likely occurs in the same accessible, arable areas. Total C sequestration was summed across each climate/ecological system for each subregion. Subregions were then aggregated to Witzke regions as described above for soil carbon.

B.3. ABANDONED AGRICULTURAL LAND

B.3.1. Location of abandoned agricultural land in Europe

Agricultural land in Europe has steadily declined over the past few decades; this basic analysis relies on the Food and Agricultural Organization (FAO) database on harvested acres and the assumption that no extensification has occurred in Europe since wide-spread agricultural abandonment began. This is likely to oversimplify the reality of a dynamic agricultural landscape, but since the effect could be either to over- or underes-timate the age of agricultural land depending on case, it is to be hoped that it does not introduce any large systematic error.

Harvested acres data were downloaded from the FAO's FAOSTAT database from 1961 to 2010 for all available countries in Europe. Turkey, Cyprus, and Russia were excluded because those countries exhibited patterns of abandonment dissimilar to the rest of Europe. The year in which steady agricultural abandonment began was identified for each country. Total harvested acres in 2010 were subtracted from total harvested acres in the starting year for each country to calculate the total area of abandoned agricultural land in each country in 2010.

B.3.2. Age of abandoned agricultural land

An age was assigned to the land abandoned in each year for each country. For example, if there were 750 hectares (ha) of land under cultivation in Iceland in 1979 and 725 ha in 1980, 25 ha of land were assumed to be abandoned for 30 years by 2010. However, if total cultivated area increased in a subsequent year, that area was assumed to be taken from the previously abandoned land, so if cultivated area was 730 ha in 1981, then only 20 ha were assumed to be abandoned for 30 years by 2010. A weighted average of age (by area) was calculated for the abandoned land for each country. Two years were then added to bring the age to the present (2012). Average ages ranged from 5 (Greece) to 39 (Netherlands), with an average of 23.7 years over all of Europe. These ages may be a slight underestimate because the data only go back to 1961, but literature review shows that this is about the time widespread agricultural abandonment began.

B.3.3. Carbon stocks on abandoned agricultural land

It was assumed that the distribution of abandoned agricultural land in Europe falls along the same biome distribution as in the WWF Terrestrial Ecoregions Map (referenced above)—this is a rough approximation. The resulting distribution of biomes among abandoned agricultural land in this analysis was 3 percent boreal forest, 29 percent Mediterranean forest/scrub, 52 percent temperate broadleaf/mixed forest, 7 percent temperate conifer forest, 8 percent temperate grassland/savanna, and 0 percent tundra.

Vegetation growth since abandonment was calculated by multiplying forgone sequestration (see the above description) by the average age of abandoned land in each country. It was assumed that for countries with an average age of abandoned land greater than 20 years, any loss in soil organic carbon (SOC) that would have occurred with cultivation had been regained and that this gain would be lost again with recultivation. Twenty years was a somewhat arbitrary time estimate but was considered conservative based on the findings in a review paper (Don, Schumacher, and Freibauer 2011) that SOC was completely restored in seven years following abandonment in tropical agricultural systems. For countries with an average age of abandonment of less than 20 years, the SOC restoration was assumed to be proportional to age. Future forgone sequestration for vegetation was calculated as detailed in the emission factors above and for SOC was assumed to be the remainder of SOC restoration for countries with average abandonment ages less than 20 years. Vegetation, soil, and forgone sequestration carbon estimates were summed.

APPENDIX C. THE ICCT SIMPLE MACRO-MODEL OF ILUC (NOSORFIM)

The ICCT's 'Not So Reduced Form ILUC Model' (refer to Plevin et al. 2010 for the "Reduced Form ILUC Model") provides an illustrative estimation of indirect land use change based on 15 parameters. These are designed to emulate macroeconomic behavior in a way that reflects more complex partial or general equilibrium models. The 15 parameters are as follows:

» E_L —The carbon emissions one would expect on conversion of the 'average' piece of land in the world to agriculture. This value is informed by data from IPCC and the Woods Hole Oceanographic Institution. The central estimate is 95 metric tons of carbon per hectare (tC/ha), with a worst case of 120 tC/ha and a best case of 70 tC/ha. This parameter captures uncertainty around the carbon lost on conversion of land in general. The distribution is characterized as normal.

» C_{yield} —The fraction of input feedstock returned to the feed market as co-products. The co-product yields vary by feedstock and are based on data used in the European Union's Renewable Energy Directive. The distribution is currently characterized as beta-PERT, as it is assumed that very high or low values are physically implausible for this parameter.

» C_{feed} —The feed efficiency of co-products. This is a characterization of the nutritional value to animals of the co-product divided by the nutritional value of the feedstock product (so a value > 1 implies that 1 metric ton of co-product would displace more than 1 ton of the feedstock in the feed market). The central value is set as one, with a lower bound of 0.8 and upper bound of 1.2. The central value is based on Klasing's (2012) and Hazzledine et al.'s (2011) work for the ICCT, the upper value on the Arora, Wu, and Wang study (2008) for the U.S. Department of Energy's Argonne National Laboratory, and the lower value is symmetric with the upper value. The distribution is currently characterized as Beta-PERT, as it is assumed that very high or low values are physically implausible for this parameter.

» C_{land} —The land carbon efficiency of co-products. This reflects the possibility that, by displacing a product other than the input feedstock, the co-product may deliver more or less carbon saving than if the net effect were to reduce total demand for the input feedstock. It is more or less the carbon saving actually achieved by the co-product divided by the carbon saving from displacing only the input feedstock. The central value varies by feedstock, with a log-normal distribution to reflect the possible long right tail of displacing very carbon-intensive products.

» σ_D —Elasticity of demand. This is the fractional change in demand that would be expected from a 100 percent change in price. The central value is set to 0.06 based on Roberts and Schlenker (2010). The distribution is set as normal, with a standard deviation of 0.025.

» σ_S —Elasticity of supply. This is the fractional change in supply that would be expected from a 100 percent change in price. The central value is set to 0.1 based on Roberts and Schlenker (2010). The distribution is set as normal, with a standard deviation of 0.04.

- » $R_{Y:A}$ —Ratio of yield to area elasticity. The ratio of yield response to price to area response to price, so a ratio of 0 means there would be no yield response. The central value is set to 0.25 based on Berry and Schlenker (2011). The distribution is log-normal, as the value could be no lower than 0, but in extreme cases, where yield response is dominant, the ratio could be very high.
- » R_M —Ratio of marginal to typical yield. This is feedstock specific, which assumes that the prevailing marginal yield behavior for a feedstock will be the behavior in the dominant producer region (e.g., sugarcane is given a marginal yield parameter of 0.9 to characterize the developing world, while wheat has a ratio of 0.85 for the developed world). The distribution is a reversed log-normal, with a firm maximum set at 1 (based on the argument that it is implausible that new land will be systematically higher yielding than existing land). These are based broadly on Keeney (2010), with a worst case from Edwards, Mulligan, and Marelli (2010).
- » R_Y —Ratio of typical feedstock yield to the actual yield achieved by the average crop that expands. This represents the possibility that for various reasons expansion could favor either lower or higher than average yield areas and crops (e.g., if sorghum for ethanol were largely replaced by U.S. corn, this would represent a higher yield than a typical U.S. sorghum yield). The central value is set to 1, with a log-normal distribution currently located at 0.75. The central value has been compared with outputs from MIRAGE.
- » F_{CS} —Fraction of land saved by general systemic efficiency improvements. This allows for adjustment across the agricultural system to improve productivity, beyond price-induced yield change and the yields of the crops that actually expand—for instance, if higher-yielding crops displace lower-yielding crops. This may have food quality implications, but those are not characterized. The central estimate is 0.15 (i.e., a 15 percent reduction in net land use change thanks to systemic efficiency gains). The distribution is log-normal located at 1, and the central value is informed by MIRAGE modeling and an unpublished Edwards et al. analysis of food quality implications.
- » R_{EF} —Ratio of emissions factors on land where expansion occurs to world average. This characterizes the possibility that expansion will systematically target lower or higher than average carbon stock land types. This is feedstock specific, but generally it is assumed that expansion will be low carbon. The distribution is log-normal, allowing for particularly high emissions when very high-carbon land types are targeted. The values are informed by several previous modeling exercises.
- » L_P —Fraction of expansion affecting peat. This varies by feedstock based on the values in MIRAGE modeling for the EU. The parameter is log-normal.
- » E_P —Peat emissions per hectare per year. The central value of 95 metric tons of carbon dioxide equivalent per hectare per year is based on Page et al. (2011a). Distribution is normal, with a standard deviation of 15.
- » Loc_{exp} —Location of crop expansion. This modifies the central estimate for the yield on expansion acreage from a typical yield for the crop category (cereals, sugars, oilseeds) to the local average yield for the input feedstock. Zero is the worldwide category average, 1 the local typical yield.

The mean values for log-normal distributions are calculated assuming that central values reflect the median.

The indirect land use change for each feedstock is calculated from these parameters as follows. First, the gross feedstock demand in metric tons of feedstock per ton of biofuel demand is calculated

$$F_{gross} = \frac{1}{Y_{Biofuel}} \tag{1}$$

for which the implied baseline ILUC emissions are calculated assuming the entire additional supply of feedstock is supplied by area expansion at a worldwide typical yield at world average land carbon

$$IF_{initial} = \frac{44}{12}\frac{E_L}{T_A}\frac{L_{gross}}{Y_{feedstock}Y_{Energy}} \text{ x } 1{,}000{,}000 \tag{2}$$

where T_A is the amortization period in years, Y_{Energy} is the energy yield from the biofuel in megajoules per metric ton, $Y_{Biofuel}$ is the yield of biofuel from feedstock in metric tons per hectare, $Y_{feedstock}$ is the feedstock yield in metric ton per hectare per year, 44/12 converts C to CO_2 , and 1,000,000 converts from tons to grams of carbon.

The net feedstock demand is first adjusted to account for co-products.

$$F_{\text{->co-product}} = F_{gross}\left(1 - C_{yield}C_{feed}\right) \tag{3}$$

Then, the feedstock supplied by demand reduction is removed, to leave only the supply response.

$$F_{supply} = F_{\text{->co-product}}\frac{E_S}{E_S + E_D} \tag{4}$$

From this, one can remove the fraction of feedstock delivered by systemic efficiency improvements from crop switching.

$$F_{net} = F_{\text{-> supply}}\left(1 - F_{CS}\right) \tag{5}$$

The net feedstock demand is then converted to net land demand at the average yield for the given feedstock.

$$L_{\text{->efficiencies}} = \frac{F_{net}}{Y_{feed}} \tag{6}$$

Next, an adjustment is made to account for the difference between the average feedstock yield and the average yield actually achieved when the cultivated area expands

$$L_{expandingcrops} = L_{\text{->efficiencies}}\,R_Y \tag{7}$$

followed by an accounting for the difference between average and marginal yield.

$$L_{net} = \frac{L_{expandingcrops}}{R_M} \tag{8}$$

From the net land demand, the ILUC in metric tons of CO_2 per ton of biofuel if all expansion occurs at world average land (soil) carbon is calculated

$$I_{averageland} = L_{net} \frac{E_L}{T_A} \frac{44}{12} \tag{9}$$

and then the actual typical carbon content of the land used for expansion is taken into account

$$I_{actualland} = I_{averageland} R_{EF} \tag{10}$$

before peat emissions are added and organic soil emissions for the equivalent area removed

$$I_{incl.peat} = I_{actualland} + \left(E_P - 10\frac{R_{EF}}{T_A} \frac{44}{12}\right) F_P L_{net} \tag{11}$$

where mineral soils are assumed to average 10 tC/ha.

Next, a co-product carbon credit is added if the co-product is more carbon efficient than the feedstock

$$I_{\text{->co-productcredit}} = I_{incl.peat} - \frac{\left(F_{gross} - F_{\text{->co-products}}\right)}{2F_{gross}\left(I_S + I_{incl.peat}\right)} \left(C_{land} - 1\right) \tag{12}$$

where I_S is some characterization of ‘normal’ ILUC emissions in metric tons of CO_2e per ton of fuel, here set to 1 ton per ton. This term allows the co-product credit to be scaled down for a parameter set that gives low ILUC while allowing that the ILUC emissions of the displaced product may not scale with the ILUC emissions of the input feedstock.

The ILUC emissions per metric ton of biofuel are finally converted to the more typical unit of grams per megajoule.

$$IF = 1{,}000{,}000 \frac{I_{\text{->co-productcredit}}}{Y_E} \tag{13}$$

THE INTERNATIONAL COUNCIL
ON CLEAN TRANSPORTATION

BEIJING BERLIN BRUSSELS SAN FRANCISCO WASHINGTON

www.theicct.org