Biofuels have gained increasing attention as an alternative to fossil fuels for several reasons, one of which is their potential to reduce the greenhouse gas (GHG) emissions from the transportation sector. Recent studies have questioned the validity of claims about the potential of biofuels to reduce GHG emissions relative to the liquid fossil fuels they are replacing when emissions owing to direct (DLUC) and indirect land use changes (ILUC) that accompany biofuels are included in the life cycle GHG intensity of biofuels. Studies estimate that the GHG emissions released from ILUC could more than offset the direct GHG savings by producing biofuels and replacing liquid fossil fuels and create a ‘carbon debt’ with a long payback period. The estimates of this payback period, however, vary widely across biofuels from different feedstocks and even for a single biofuel across different modelling assumptions. In the case of corn ethanol, this payback period is found to range from 15 to 200 years. We discuss the challenges in estimating the ILUC effect of a biofuel and differences across biofuels, and its sensitivity to the assumptions and policy scenarios considered by different economic models. We also discuss the implications of ILUC for designing policies that promote biofuels and seek to reduce GHG emissions. In a first-best setting, a global carbon tax is needed to set both DLUC and ILUC emissions to their optimal levels. However, it is unclear whether unilateral GHG mitigation policies, even if they penalize the ILUC-related emissions, would increase social welfare and lead to optimal emission levels. In the absence of a global carbon tax, incentivizing sustainable land use practices through certification standards, government regulations and market-based pressures may be a viable option for reducing ILUC.

Keywords: indirect land use change; economic models; environmental externalities

1. INTRODUCTION
Biofuels have gained increasing attention as an alternative to fossil fuels for several reasons, one of which is their potential to reduce the greenhouse gas (GHG) emissions from the transportation sector. The biofuels industry in the United States (US), the European Union (EU), Brazil and increasingly in Southeast Asia is currently based on technologies that use food/feed crops as feedstocks. With the exception of Brazil, biofuel production in other countries is competitive with liquid fossil fuel only at high oil prices or with government support. Governments across the world, including the US, UK and the EU, have established several policies to support a transition away from liquid fossil fuels to low carbon fuels, including mandates and subsidies for blending biofuels with liquid fossil fuels and low carbon fuel standards. The objectives for encouraging biofuel production are diverse and include increasing energy security, stimulating rural development and supporting a transition to a low carbon economy, in addition to climate change mitigation. The implementation of some of these policies relies on a life cycle assessment of the GHG emissions intensity of the biofuel and has led to a controversial debate about the components of the life cycle assessment that should be considered, and the validity of claims about the potential for biofuels to reduce GHG emissions relative to the liquid fossil fuels they are replacing.

GHG emissions from the production of biofuels depend on energy/carbon inputs used directly over

*Author for correspondence (khanna1@illinois.edu).

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the life cycle of the production of biofuels, as well as on the carbon emitted (or sequestered), as land is converted from existing uses to biofuel feedstock production. Emissions from direct land use change include those from changes in the carbon stocks in soil and vegetation. The diversion of globally traded food/feed crops for biofuel production and the competition for cropland induced by biofuel production has the inevitable impact of raising world prices of not only the biofuel feedstocks but also of other crops that compete for limited land resources. The increase in world prices could induce crop acreage expansion on native vegetation and forested land in other regions, which releases the carbon stored in these ecosystems, leading to ILUCs that also contribute to GHG emissions (figure 1).

An assessment of the ILUC effect on GHG emissions is important for determining the extent to which biofuels are reducing global GHG emissions from transportation fuels. However, unlike direct emissions from fossil fuel use that can be physically measured or calculated using process-engineering methods, the ILUC effect of biofuels has only been determined so far, using economic models that can be used to quantify the GHG effects of land use changes (LUCs).

Studies estimate that the GHG emissions released from ILUC accompanying biofuel production could more than offset the direct GHG savings by producing biofuels and replacing liquid fossil fuels and create a ‘carbon debt’ with a long payback period [1–3]. The estimates of this payback period, however, vary widely across different types of biofuels [4,5]. First-generation biofuels are produced from food-based feedstocks such as corn, wheat and soya beans and have larger ILUC-related GHG emissions than second-generation biofuels that are expected to be produced from dedicated energy crops and crop and forest residues. The estimate of the ILUC effect also varies for a single biofuel across studies owing to differences in the structure of the model and its parametric assumptions and within a study (as assumptions are changed). For example, the payback period for corn ethanol is estimated to range from 15 to 200 years across studies [5,6]. Inherent differences among modelling approaches, the need for numerous assumptions to support the forward looking analysis required to determine ILUC effects and limited empirical data to support or refute particular assumptions make it unlikely that these models will lead to identical results about the ILUC effects of a biofuel [1].
determination of a unique ILUC factor is made even more difficult by the fact that ILUC effects are sensitive to policy choices. As biofuel production is primarily policy-driven in the US and EU, the mix of policies and the magnitude of the policy shock influence the nature and size of the ILUC effect.

While some have argued against the inclusion of ILUC effects for reasons including the large uncertainties associated with measuring them [7,8], others argue against including a value of zero in policy implementation because the ILUC emissions for corn ethanol are always larger than zero (despite the uncertainty) and a zero value would lead to perverse outcomes [6]. The US Environmental Protection Agency (EPA) recognizes the uncertainty in estimating the ILUC effect of the Renewable Fuels Standard (RFS) and quantifies it using confidence intervals to determine if specific types of biofuels meet its threshold requirements for GHG intensity reduction relative to petrol [4,9].

To illustrate the challenges in estimating the ILUC effect, we describe some of the key assumptions made in these different models and the sensitivity of their results to these assumptions about parameters and policy scenarios. In doing so, we consider models whose features and sensitivities have been analysed in the literature and those used by policy-makers in the US and EU to determine the GHG impact of biofuel policies and ILUC in particular. These models differ in the types of biofuels they incorporate. The Global Trade Analysis Project (GTAP) and Food and Agricultural Policy Research Institute (FAPRI) models consider only first-generation feedstock from food/feed crops [10,11], while the Food and Agricultural Sector Optimization Model (FASOM) includes several second-generation biofuel pathways as well [12]. The GTAP model has been used to analyse the ILUC associated with California’s low carbon fuel standard (LCFS), while FAPRI and FASOM have been used to analyse the impact of the RFS in the US. In the EU context, the Modelling International Relationships in Applied General Equilibrium (MIRAGE) model is used to examine effects of DLUC and ILUC of the EU renewable energy directive (RED) and considers various first-generation feedstocks for ethanol and biodiesel [13]. Other models such as the Integrated Global Systems Model (IGSM) have been used to simulate land use effects of a global climate policy, with a generic second-generation biofuel as an alternative to petrol [14]. Discussion about other models used to evaluate the land use impact of biofuels can be found in Prins et al. [5] and Witzke et al. [3] and Fonseca et al. [1].

Other approaches that rely on a ‘causal–descriptive’ methodology are currently being developed by the UK Department for Transport to address ILUC of biofuels derived from a limited number of feedstocks [15].

In addition to the difficulties in estimating ILUC and uncertainties about the correct estimate of the ILUC factor for policy implementation, the presence of ILUC raises both normative questions and empirical questions about policy choices and their implications. A key normative question is whether the GHG emissions owing to ILUC should be included in regulatory implementation. Based on principles of environmental economic theory, Zilberman et al. [7,8] argue against the idea of penalizing biofuel producers for market-mediated externalities that these producers do not control. Instead, they emphasize the need for a global carbon tax or cap-and-trade policy that prevents these leakages of emissions in the first place. The carbon price under these policies would be set equal to the monetary value of the environmental damage caused by the last unit of GHG emissions emitted or to the market price of carbon which ensures that an emissions cap is met. Such a policy would incentivize mechanisms for reducing global GHG emissions at least cost.

Difficulties in implementing a global carbon policy or even national carbon policies have led some countries to develop alternative policies to mitigate GHG emissions. These policies include biofuel mandates, subsidies and GHG intensity standards and have led to wide recognition of the problem of GHG emissions leakages owing to ILUC. Concerns that these indirect emissions may negate the GHG motivation for biofuels have led regulatory agencies in the US to include GHG emissions caused by ILUC in the life cycle assessment of GHG emissions intensity of biofuels, when designing biofuel regulations. In the EU, concerns about indirect emissions have led the ongoing studies to assess modelling methodologies and their suitability for use in reporting ILUC factors and discussions about ways to deal with ILUC in a regulatory framework are ongoing.

In the absence of policies that directly penalize GHG emissions across the globe, the presence of ILUC-related GHG emissions that accompany these alternative policies raises several questions of an empirical nature. What are the economy-wide costs and implications for global GHG emissions of including the ILUC effect in these policies? What is the trade-off posed by the inclusion of the ILUC factor in policy implementation between the various objectives that motivate biofuels? Are there other alternative policy approaches for avoiding or minimizing ILUC more effectively? We discuss the implications of ILUC for the potential for biofuels to be one solution to climate change mitigation and the policies that need to accompany them to realize that potential. The rest of this paper is organized as follows. Section 2 describes the different types of biofuels and their GHG intensities. Section 3 discusses the biofuel policies in the major biofuel producing/consuming regions of the world where concerns about ILUC may have been paramount. Section 4 describes some of the key modelling assumptions needed to measure ILUC. Section 5 describes the sensitivities of ILUC estimates to the structure and parametric assumptions of the models used. Section 6 discusses the implications of these findings for policy design and §7 concludes.

2. THE GREENHOUSE GAS INTENSITY OF BIOFUELS FROM ALTERNATIVE FEEDSTOCKS

The extent to which biofuels can mitigate climate change depends on their GHG intensity relative to the liquid fossil fuels they displace. This GHG intensity is
sensitive to the practices used for producing the feedstock and the biorefinery heat source. Farrell et al. [16] show the wide range in estimates of the direct GHG intensity of corn ethanol, in part, owing to differences in input application rates. Liska et al. [17] show that the direct GHG savings from corn ethanol relative to petrol range between 17 and 59 per cent depending on whether coal or biomass is used as the source of energy for the biorefinery. GHG intensity of biofuels also differs across first- and second-generation biofuels and the feedstocks used for each. Studies consistently show that second-generation biofuels from cellulosic feedstocks, particularly perennial grasses like Miscanthus and switchgrass, have significantly lower direct GHG emissions intensity than corn ethanol or petrol [16,18–20]. Farell et al. (2006) estimate that corn ethanol can reduce GHG emissions by 18 per cent, while ethanol from switchgrass is expected to reduce emissions by 88 per cent relative to petrol. The emission intensity of sugar cane ethanol is also estimated to be significantly lower than that of corn ethanol [21].

DLUC and ILUC effects of biofuels also differ considerably across feedstocks. Perennial grasses or energy crops like Miscanthus and switchgrass are relatively higher yielding than other biofuel feedstocks and have the potential to sequester significantly large amounts of carbon on previously cultivated land [22–25]. They can also be grown productively on low-quality marginal lands [26]. Thus, the DLUC from existing uses, either crop production or marginal pastureland, to perennial grasses, can make biofuels from these grasses a net sink of carbon rather than a source of carbon. The ILUC effect of cellulosic biofuel from energy crops is also likely to be low because a unit of this biofuel requires much less land than a unit of corn ethanol, as ethanol yields of energy crops could be much higher than those of corn-based ethanol even after considering the co-products produced by the latter. Moreover, these grasses are more likely to be economically viable on marginal land because of its lower opportunity costs relative to other cropland; hence they are much less likely to displace food/ feed production and adversely affect world market prices when compared with food crop-based biofuels (see [27]). High oil prices or large subsidies for energy crops could, however, make these grasses competitive on cropland and provide incentives to convert high-quality cropland as well as native grasslands (despite their costs of conversion) to bioenergy production. Policy choices are, therefore, needed about the extent to which expansion of energy crops should be encouraged and the types of land on which they can be grown. Cellulosic biofuels can also be produced from crop or forest residues, which would have negligible ILUC effects as they are a by-product of other production activities; however, the removal of these residues does have adverse consequences for soil organic carbon, which reduces their direct potential for mitigating GHG emissions [22].

A commercial technology to produce second-generation biofuels from cellulosic feedstocks is currently not available. The process of converting these feedstocks to biofuels is currently significantly more expensive than first-generation biofuels [19,28]. These costs can be expected to decline with up-scaling of production from pilot scale to industrial scale, learning by doing and cumulative experience. There is empirical evidence of cost reductions in the production of corn ethanol and sugar cane ethanol due to learning by doing [29,30]. De Witt et al. (2006) develop learning curves for first- and second-generation biofuels and show that the potential to reduce the conversion costs for advanced biofuels between 2005 and 2030 is considerably larger for advanced biofuels than for first-generation biofuels. The costs of first-generation biofuels are expected to stabilize while those of second-generation biofuels can be expected to decline by more than 30 per cent over the next 25 years, making them cost competitive with liquid fossil fuels if priced in the range of $70–130 per barrel. Government policies, like biofuel mandates and investment subsidies, which stimulate investment in the production of these biofuels can play an important role in accelerating their production and lowering costs of production in the future.

3. BIOFUEL POLICIES IN THE EUROPEAN UNION, UNITED KINGDOM AND THE UNITED STATES

There are various motivations for the policies promoting renewable energy in the US and EU, including energy security, reducing GHG emissions, stimulating innovation that reduces dependence on exhaustible oil and leads to a low carbon economy in the future, and rural economic development. The increased use of biofuels as a source of liquid transport fuel is being required by various laws and directives in the EU member states, the UK and the US. The need to increase the share of liquid biofuels for transport was recognized in the EU as a means of addressing future long-term fossil resource depletion and as a means to reduce GHG emissions. Biofuels have been promoted by various policies, including the Fuel Quality Directive (FQD) and as part of the RED [31,32]. The FQD aims to reduce life cycle GHG emissions from fuels consumed in the EU by 6 per cent by 2020.

1First-generation biofuels here refer to existing conversion technologies such as fermentation of sugars and starches to produce ethanol (derived from commodity food crops such as maize, sugar cane, wheat and sugar beet) and methylesterification of vegetable oils to produce biodiesel (derived from commodity crops such as oilseed rape/canola, soya and palm oil). Second-generation refers to advanced biofuel technologies such as lignocellulosic ethanol (derived from crop residues or dedicated energy crops such as energy cane, switchgrass, Miscanthus, short rotation coppice species).

2The ILUC effect per unit of these biofuels produced on cropland would still be lower than that of food-based biofuels because of their higher yields per unit land, as stated above.

The European Commission states in its Communication from the European Commission: an EU Strategy for Biofuels, that ‘the EU is supporting biofuels with the objectives of reducing GHG emissions, boosting the decarbonization of transport fuels, diversifying fuel supply sources and developing long-term replacements for fossil oil’, adding also that ‘the development of biofuel production is expected to offer new opportunities to diversify income and employment in rural areas’ (COM (2006) 34 (8 February 2006), 3).
The RED sets a 10 per cent binding target for renewable energy use in road transport fuels and an overall binding target to meet 20 per cent of the EU energy needs from renewable sources by 2020. As part of the overall target, each member state has to get at least 10 per cent of their transport fuel consumption from renewable sources (including biofuels). The RED and the FQD provide sustainability criteria for biofuel production and set standards for the land used to grow feedstock. ILUC is currently not included in the GHG calculation made for both the FQD and the RED, but there is ongoing review of models that might be used to define ILUC. Both the FQD and the RED are now component parts of the EU Energy and Climate Change Package, which became law in June 2009. Under this legislation, 27 EU Member States committed to reduce CO₂ emissions by 20 per cent by 2020 and to target a 20 per cent share of renewable energies in EU energy consumption by 2020, i.e. ‘20-20-20’ delivery targets.

In the UK, biofuels have been promoted since 2001, under the green fuel challenge that led to a reduction in import duties for biodiesel fuel, which at the time was derived from used cooking oil. Since then, the renewable transport fuels obligation (RTFO) has been developed, with GHG emission outcomes as a key component and it has therefore been necessary to stipulate the methodology and processes required to report GHG emissions from the individual biofuel supply chains used by obligated parties in law [33]. The RTFO specifies a methodology for assessing carbon intensity and sustainability of biofuel supply chains from feedstock source, by country and by on-farm production inputs and outputs. It currently does not include ILUC, but the inclusion of an ILUC factor is under review.

Unlike the EU and the UK, biofuels policy in the US has been primarily motivated by energy security concerns that led to the Energy Independence and Security Act (EISA) of 2007 and the RFS that aims to increase the volume of renewable fuel from 34 billion litres in 2008 to 136 billion litres by 2022 [4]. GHG mitigation was viewed as a secondary benefit of biofuels. The RFS sets volumetric requirement for different types of biofuels based on their life cycle GHG intensity including ILUC effects and caps corn ethanol production at 56 billion litres from 2015 onwards to encourage a transition to cellulosic biofuels. The RFS stipulates that biofuel from new corn ethanol refineries should achieve a life cycle GHG emission displacement of 20 per cent compared with petrol in order to qualify as conventional biofuels. The rest of the mandate has to be met by advanced biofuels produced from feedstocks other than corn starch that achieve a life cycle GHG emission displacement of 50 per cent when compared with the petrol they displace. Of this at least 60 billion litres must be from cellulosic feedstocks that achieve a life cycle GHG emission displacement of 60 per cent when compared with the petrol they displace. These life cycle emissions include those owing to ILUC effects. While volumetric mandates for cellulosic biofuels have been set from 2010 onwards, the EPA has the authority to revise these in the absence of commercially viable production. The targets for 2010 were, therefore, scaled back considerably [34]. The California Environmental Protection Agency Air Resources Board (CARB) has been at the forefront of developing policy to reduce transportation section emissions and has developed the LCFS, which calls for a 10 per cent reduction in the GHG intensity of transport fuels by 2020 [35]. California’s regulations require a gradual reduction in the carbon intensity of the fuel sold in the state. It encourages the consumption of not only biofuels but also other alternative fuels, such as natural gas and electricity. Assuming that the California LCFS will not lead to any incremental biofuel production beyond that required to meet the RFS, CARB’s estimates of the GHG intensity of biofuels include their ILUC effect, measured using the GTAP model. The RFS and the LCFS differ in the extent to which they incentivize low GHG intensity biofuels. Under the RFS, a particular biofuel pathway only has to meet a certain threshold of reduction in GHG intensity to be qualified to meet the mandate. By treating all cellulosic biofuels the same, the RFS does not give incentives to use feedstocks that can lead to even lower GHG intensity than 60 per cent. It also does not limit imports of unconventional crudes with higher GHG intensity than petrol. In contrast, the exact GHG intensity is important for the LCFS, as the carbon intensities of fuels in the fuel mix are averaged to evaluate whether the entire fuel supply adheres to the carbon intensity standard. The LCFS creates incentives for continuous improvement in lowering the GHG intensity of biofuels. Moreover, it implicitly penalizes the use of high-carbon fuels like unconventional crudes and thus reduces incentives to import them. While the California LCFS may not lead to additional ILUC effects beyond the RFS, it could create other leakages by diverting unconventional crudes from Canada, destined for the US, to the rest of the world.

4. DETERMINING THE GREENHOUSE GAS INTENSITY OF BIOFUELS WITH LAND USE CHANGE

Figure 1 illustrates the pathway (in the blue boxes) through which a policy change affecting biofuel production affects land use domestically and in the rest

Interface Focus (2011)
of the world, leading to GHG emissions from the change in land use, which could more than offset the GHG savings from substituting biofuel for petrol. A policy-induced increase in biofuel production would increase the demand for biofuels crops and divert cropland used for food/feed or marginal pastureland to biofuel feedstock production (DLUC). These DLUCs lead to GHG emissions if they lead to the expansion of biofuel crops on virgin grassland or forest land and to GHG sinks if they lead to the production of energy crops with a high potential to sequester soil carbon on cropland with low levels of accumulated soil carbon. The increased competition for land would increase the prices of not only biofuel feedstocks but also other crops competing for land inputs. If the country is a large trader in the world market, world prices of commodities will be affected, leading producers to intensify production on the existing cropland and to bring new land into cultivation. The latter are ILUCs caused by biofuels that can lead to GHG emissions, depending on the type of land that is converted to crop production and the carbon that is stored in soil and vegetation on that land.

The determination of the ILUC effects of biofuel production requires simulation of behaviour not only in the biofuel producing country but also the transmission of impacts on world markets and on the behaviour of land owners and food consumers in all regions of the world.

4.1. Emissions

Depending on the structure of the simulation model, this typically requires assumptions about growth in crop productivity, the ease of conversion of land across uses, the availability of non-cropland for conversion and its crop productivity, and the ease of substitution between imported and domestically produced commodities as indicated in the green boxes in figure 1. These assumptions are likely to differ across the different regions of the world. We now discuss the need for these assumptions as well as the ways in which they affect the outcomes of the models.

Decisions about how to use agricultural cropland are made by landowners and economic analysis of the factors driving these decisions is based on the premise that land is used in the activity that generates the highest rents [36]. Agricultural landowners need to decide on the mix of crops to grow, the management practices and technology to adopt, and the amount of their land to cultivate or leave idle. These decisions are likely to be influenced by expectations about output and input prices, soil productivity and climatic conditions. Land that is less productive may be left idle when crop prices are low and brought into production when crop prices increase. As land is heterogeneous in its productivity, location to markets and climate conditions, decisions about land use differ spatially within and across countries. These decisions can change over time in response to exogenous policy shocks, technological change, and changes in domestic and world market conditions.

Models that seek to quantify the land use implications of biofuels and simulate the effects of biofuel policies need to integrate across many different scales. A global representation of agricultural and forestry markets is needed to capture the effect of biofuel-induced changes in land use and prices on international trade and land use in other countries. At the same time, the assessment of biofuel impacts requires a high degree of spatial resolution to account for heterogeneous land qualities, climate, land availability and ease of its conversion from one use to another. These assessments need to be based not only on models that capture the economic behaviour of agents but also the technology of crop production, the biophysical factors that affect crop productivity, and the land suitability and availability constraints on cropland expansion. Models capture these heterogeneous individual behaviours by aggregating individual decisions within a homogeneous simulation unit (i.e. a unit of land which could be a county, country or continent). The extent to which biophysical considerations are incorporated and the size of these simulation units differs greatly across models [5].

Several types of economic models are being used to study the land use effects of biofuels. These can be classified into dynamic partial equilibrium models, static partial equilibrium models, and dynamic and static general equilibrium models. Partial equilibrium models focus on a few sectors of the economy that are most closely associated with biofuel production, namely the agricultural, forestry and fuel sectors. Prices, production and land allocation within these sectors are determined within the model and it is assumed that conditions in the rest of the economy and the prices of labour and capital will remain unchanged with biofuel production. Dynamic models examine decisions over a long time horizon and decisions in one period affect those in the next period. In contrast, static models examine the effects in a single period of time. Some of the models whose results are discussed below are described here briefly to illustrate significant differences in their structure and the mechanism that drive $\Delta$LUC and ILUC in response to biofuel production.

Unlike partial equilibrium models, computable general equilibrium (CGE) models simulate economy-wide effects of a biofuels shock and include inter-sectoral linkages and constraints on labour and capital, and determine all prices in the economy simultaneously. These models are global in scope and represent multiple economic sectors and include factor markets for labour and capital. While broad in geographical and sectoral scope, many CGE models have limited spatial resolution and partition the world into a few large homogeneous regions called agro-ecological zones. Each region has a regional representative household that allocates resources domestically. A production sector produces goods and services using consumer-owned endowments as primary inputs. Each region interacts with other regions through trade. Consumers maximize utility, while producers maximize profits, leading to endogenously determined prices and quantities of goods and factors of production. These models also limit the number of agricultural products considered by categorizing individual commodities into large groups (e.g. all coarse grains) and imposing
the same behavioural and market assumptions on the individual components.

Dynamic partial equilibrium models like FASOM are multi-period, multi-commodity, multi-factor market models [12]. A key feature of these models is that they assume inter-temporal optimizing behaviour by economic agents and are well suited to endogenously determine the least cost mechanisms and mix of feedstocks to meet policy-induced biofuel targets, while constraining the total amount of land available. Landowners are assumed to choose allocation of land among alternative uses based on the net present value of the future returns to alternative activities, subject to some constraints that prevent large deviations from historically observed land use patterns. FASOM includes several types of first- and second-generation biofuels. It cannot however be directly used to estimate ILUC as it lacks the regional detail at the global level.

ILUC effects of the RFS are estimated by running FASOM in parallel with the FAPRI model, which is a static multi-market, partial equilibrium model with a more detailed representation of international land use. FAPRI comprises a system of supply and demand equations for many related markets across the globe, which are solved simultaneously to determine the equilibrium market prices that clear supply and demand in all markets at a point in time [10]. In a multi-market model like FAPRI, biofuel production is induced by applying an exogenous shock to the system, such as a rise in the price of biofuels or an increase in the quantity demanded of biofuels that violates the supply and demand equilibrium and then iteratively solves the system until the markets are once again in equilibrium. Land constraints are not explicitly included in the FAPRI model and the model currently considers only food/feed crops as the source of first-generation biofuel feedstock.

General equilibrium models include the IGSM, the GTAP model and the MIRAGE model [11,13,14]. All three models are using essentially the same database (supplied by GTAP) but differ in the base year used for calibration. IGSM and MIRAGE are both dynamic models, with varying time steps and time horizons, while the GTAP version used for the purpose of analysing the LCFS is an intrinsically static model [35,37]. IGSM and MIRAGE are different from GTAP, which considers only managed land, in that they model the conversion of natural forests and grasslands into crop-land or pasture land. The IGSM framework has the most sophisticated emissions modelling, as it features full dynamic accounting of carbon fluxes in vegetation and soils. In contrast, GTAP and MIRAGE use constant factor intensities of conversion from one land use to another. However, the modelling of biofuel production pathways is relatively coarse in the IGSM, while GTAP and MIRAGE have fairly detailed modelling of first-generation biofuel production pathways, including co-products and interaction with the livestock industry. The three general equilibrium models feature differentiated land quality based on productivity, climatic conditions and access. However, the resolution is different; GTAP and MIRAGE use 18 agro-ecological zones that differentiate land into six categories of 60 day growing period intervals and three climate types. The IGSM model divides global land cover into $0.5^\circ \times 0.5^\circ$ grid cells.

5. Sensitivity of indirect land use change to modelling assumptions

We now describe some of the assumptions made by economic models analysing the land use and GHG impact of biofuels and discuss sensitivity of the ILUC estimate to those assumptions.

5.1. Policy simulation

The policy or set of policies being modelled, the size of the policy shock considered and the counterfactual baseline used for comparison affects the magnitude of the resulting DLUC and ILUC. Searchinger et al. [2] simulate an increase in corn ethanol production of 56.7 billion litres, above the reference case which already projects almost 56.7 billion litres of corn ethanol. Thus, the implied total biofuel production is 113.4 billion litres. This policy mix is likely to lead to fewer exports of corn and a larger corn price increase, resulting in a higher ILUC effect than studies by Hertel et al. [37] and Tyner et al. [38] that simulate a smaller shock of only 50 billion litres increase in corn ethanol production over the baseline production of 6.7 billion litres. In simulating the effect of the RED using MIRAGE, Al-Riffai et al. [13] find that if the policy is accompanied by trade liberalization, the ILUC effect increases owing to the shifting of production from the EU to countries like Brazil, which may be more prone to convert native vegetation to crop-land. However, emissions from production and DLUC decrease owing to greater production efficiencies in those countries. As a result, the net emission savings (compared with petrol) after considering DLUC and ILUC emissions increase by 8.7 per cent. They also show that a 50 per cent increase in biofuel production from 35.5 to 54.4 billion litres almost doubles the carbon intensity of biofuel from ILUC from 17.7 to 30 g CO2 per MJ because it leads to the use of less productive new land. Tyner et al. [38] examine the ILUC effect of an increase in biofuel production to 56.7 billion litres in 2015 relative to several baselines: a 2001 baseline, a 2006 baseline and a 2006 baseline with population and yields allowed to increase over the period 2006–2015. They find a difference of 50 per cent in the ILUC effect (14–21 g CO2 per MJ), with the upper and lower bound corresponding to the first and third set of simulations.

5.2. Yield responsiveness

Some of the key parameters that influence the magnitude of the ILUC factor are those that determine the intensiveness with which cropland is used and the productivity of changes in land at the extensive margin. Additionally, assumptions about the rate of growth of crop productivity and the productivity of marginal land that is brought into production are found to
have a significant impact on the ILUC effect. As crop prices increase, crop producers are induced to bring less-productive or idle land into production. However, the productivity of new land brought into production is likely to be lower than that of the existing cropland. The lower the productivity of marginal land in biofuel-producing countries, the greater the increase in crop prices. The lower the productivity of marginal land in other countries in the rest of the world, the greater the ILUC effect in response to crop price increases. These parameters are likely to differ across different regions in the world.

Studies differ in their assumptions about the yield of biofuel feedstocks. Searchinger et al. [2] assumed a corn ethanol yield of 3766 l ha⁻¹; Hertel et al. [37] assumed 3598 l ha⁻¹ while the EPA assumed that corn ethanol yield will reach nearly 4423 and 4692 l ha⁻¹ in 2017 and 2022, respectively (see [6]). Furthermore, some studies assume responsiveness of crop yields to technological changes over time and to crop prices [27,38], while others include only a time trend for crop yields [2,12] or allow yields to be responsive to prices only [37].

Searchinger et al. [2] assume that yields increase according to observed trends but that the yield increase is offset by reduction in productivity owing to expansion of production on marginal lands. Their estimate of a payback period for corn ethanol of 167 years reduces to 31 years if the assumed rate of growth in yields is 1 per cent higher over 10 years [10]. Econometric studies also show that yield is responsive to price expectations, but there is a considerable range of estimates of the price elasticity of corn yield from 0.22 to 0.76 and for coarse grains from negative to 1.5 in the long run [39]. Similarly, the estimate of the productivity of marginal land varies across studies. Keeney [40] reports that estimates of the ratio of marginal to average land productivity range from 0.47 to 0.9 in the literature. MIRAGE assumes that the productivity of marginal land is half of the average productivity in the existing cropland for all regions and 75 per cent in Brazil [13], while GTAP assumes this value to be 66 per cent [37]. Using GTAP, Keeney [40] simulates the effect of 56.7 billion litres of biofuel production in the US on ILUC in Brazil, for a range of ratios of marginal to average yields and a range of yield price elasticities in Brazil. For the range of values of the two parameters considered, the additional cropland hectares converted to crop production in Brazil range between 190 000 and 489 000. Varying the most uncertain yield parameters in the GTAP model leads to a range of 15–90 g CO₂ per MJ in the ILUC factor with a central estimate of 27 g CO₂ per MJ [37].

5.3. Land available for conversion

Models also differ in the land that is considered to be available for conversion to cropland. While FASOM and GTAP restrict land conversion to managed cropland, pastureland and forests [12,38], other models such as IGSM and MIRAGE allow unmanaged grasslands and natural forests to be converted to cropland if it is profitable [13,14]. These models do not fully account for all the costs of converting unmanaged lands to cropland and the regulations restricting conversions and may lead to higher conversion rates than observed in reality. There is some evidence that even with increasing biofuel production in the recent years, deforestation rates in Brazil are decreasing. Recent data for Brazil show that the rate at which the Brazilian Amazon is being cleared decreased drastically in 2008–2009 when compared with the previous decade, owing to government regulations to expand protected areas, policies to discourage cultivation of sugar cane in the Amazon, better policing and low commodity prices [41]. These changes in government policies are not reflected in the models attempting to measure ILUC.

Allowing the potential for conversion of unmanaged natural areas to cropland leads to a sevenfold higher ILUC effect of biofuels when compared with a scenario that allows only more intensive use of the existing management [14]. Al-Riffai et al. [13] find that decreasing the land extension elasticity (acreage elasticity of unmanaged land) in Brazil by half from 0.1 to 0.05 reduces global ILUC emissions owing to the EU RED policy by one-third. Searchinger et al. [2] estimate that 36 per cent of new cropland in the US for corn ethanol production comes from forest, while Hertel et al. [37] estimate that 19 per cent of the net conversion to cropland comes from forests and the rest from pasture. The latter is more consistent with the observed data, which suggests that most of the additional cropland in the US has come from grasslands that were in pasture and not from forests, and that forest area in the US has remained constant from 1990 to 2005. Eliminating the potential for deforestation in the US reduces the payback period in Searchinger et al. [2] from 167 years to 141 years [10].

5.4. Ease of land conversion

In many models, the ease of conversion of land from one use to another is measured by the elasticity of substitution between land and non-land inputs in the production function for individual crops, as well as the elasticity of substitution between different land classes. This introduces nonlinearity in the ease of conversion of land from one use to another; with the implicit costs of that conversion increasing as more land is converted. This reflects the fact that pasture and forestry land converted to cropland has decreasing marginal productivity and there are institutional factors that could hinder the conversion of these lands to cropland. The lower the value of this elasticity the quicker the decline in marginal productivity of 1 ha moving from one use to another and the larger the extensive margin and ILUC effect on non-cropland. Empirical evidence on the magnitude of the elasticity of land substitution and how it varies across regions and in response to particular crop prices is limited.

In other models, land conversion occurs based on the net returns to land and a conversion cost that is incurred for converting unmanaged land to managed land. This may lead to a lower elasticity of substitution across land uses in the short run but a larger value in the long run. Using MIRAGE, Al-Riffai et al. [13] find
that an increase in the elasticity of substitution between crops or between cropland and pasture by 50 per cent, decreases global emissions associated with EU RED by 10 per cent and reduces the ILUC effect by 30 per cent because the ease of converting land from one use to another reduces the need to bring in less-productive new land. Using IGSM, Gurgel et al. [42] show that global bioenergy production is 10–20% greater when land conversion is based only on net returns as compared to when it is based on the elasticity of substitution, which tends to limit market response to follow observed historical trends. Given the potential for unprecedented changes in land use owing to biofuels, the reliance on historical trends may be overly restrictive. The use of an elasticity of substitution parameter also has the disadvantage that it allows significant differences in returns to land in the same agro-ecological zone to persist over time; in the long run, it is more reasonable to expect that land would shift towards uses with relatively higher returns until returns are equalized across all uses.

5.5. Import and export responsiveness to prices

The diversion of cropland away from traditional crops to biofuel feedstock production reduces production and either reduces the supply of exports or increases the demand for imports for the affected crops. Models differ in their assumption about the responsiveness of imports and exports to price changes. Some models such as GTAP and MIRAGE place restrictions on the responsiveness of imports and exports by constraining the substitution of imported and domestically produced goods. Others such as IGSM and most partial equilibrium models, like FAPRI, assume that an imported good and its domestically produced variety are perfect substitutes and that one world price exists for that good. The later assumption allows for an easier transmission of a price shock throughout the world economy, whereas placing restrictions on the substitutability of imports and domestic goods tend to limit market response to follow the observed historical trends. As a result, the mean land conversion is smaller using the GTAP model when compared with the FAPRI model [43]. The appropriateness of assuming perfect substitution versus limited substitution for imported and domestic goods is an empirical question that is under debate. Some empirical evidence for limited substitution is presented by Villoria & Hertel [44] for coarse grains but Golub et al. [11] note that this result is not definitive and the appropriate assumption needs to be investigated for specific commodities.

5.6. Time horizon

Estimation of the effect of ILUC on the GHG intensity of biofuels per MJ requires comparing the one-time release of carbon from converted areas like forest and grasslands at the time of conversion with annual soil carbon sequestration on land converted to crops and the annual displacement of GHG emissions when biofuels replace liquid fossil fuels as a fuel. The net impact depends on the length of the time horizon assumed for the comparison and the choice of the latter is arbitrary. If one integrates over a long enough period, biofuels may show a substantial GHG advantage, but over a short period some biofuels have a higher carbon intensity than fossil fuels. Studies and policies differ in the time horizon they use for estimating the ILUC effect. The EPA uses a 30 year time horizon for calculating the ILUC factor used in the RFS. The MIRAGE model used to analyse EU RED uses a 20 year horizon, while the IGSM uses a range of time horizons, from 30 to 100 years. Assuming a 30 year time horizon for comparison results in a 50 per cent lower ILUC factor compared with that of a 20 year time horizon using GTAP [37].

Furthermore, existing studies typically assume straight-line amortization to annualize the one time indirect emission over the time horizon. This approach assumes that emissions in year 1 have the same global warming potential as emissions in year 30. O’Hare et al. [45] argue that initial emissions have a larger effect on global warming at any time in the future because they incur forcing for a longer period relative to later emissions, and proper accounting of the differential time of emissions discharge would increase land use emissions attributed to biofuel, compared to using straight-line amortization. However, assessing the effects of ILUC on GHG emissions based on their global warming potential requires assumptions about a rate of discount to value future emissions as being less damaging than immediate emissions and an assessment of the marginal damage over time which is uncertain and depends on the level of GHG emissions at that point in time [14].

There are additional factors that can cause ILUC effects to vary that were not discussed above. These include the assumed mix of available feedstocks, the co-products of biofuels, and the life cycle energy and GHG intensity of the biofuel production process. While GTAP and FAPRI consider only first-generation biofuels, FASOM allows for multiple first- and second-generation feedstocks, including crop and forest residues, to compete with each other to meet the RFS. With multiple feedstocks, attributing an ILUC effect to each feedstock requires a marginal analysis that determines the incremental contribution of a marginal increase in biofuels using one particular pathway holding the total level of production of biofuels from all feedstocks constant [43]. The mix of feedstocks included in this analysis, the relative costs of producing biofuels from crop and forest residues relative to energy crops, the assumed yields of energy crops, the type of land on which these energy crops can be grown and the carbon intensities of different biofuel pathways will influence this marginal ILUC effect of each biofuel pathway (see [5]). Additionally, most of the multi-market models described here focus on a supply side analysis of a biofuel production/price shock, assuming that the additional biofuels can substitute for petrol on an energy-equivalent basis. This ignores bottlenecks owing
to lack of infrastructure to distribute biofuels and the inability of the current vehicle fleet to accommodate its use. In addition, some models such as FAPRI ignore the feedback effects of changes in the oil price on demand for biofuels and the rebound effect of biofuels on global petroleum markets. As a result, the oil price shock considered by Searchinger et al. [2] lead to a much greater increase in biofuel production than it would if these bottlenecks are considered (as discussed in [10]). There are also substantial uncertainties in estimating the GHG emissions from LUCs over large regions. The estimation of emissions requires information on carbon stocks in vegetation and annual changes in sequestration from land use conversion [6]. Finally, the ILUC effect of biofuels depends on the policies used to support them. Policies that encourage greater production of second-generation biofuels or of biofuels with lower GHG intensity will have a smaller ILUC effect than policies that encourage the production of food-based biofuels. The ILUC effect of a biofuel is, therefore, specific to a policy mix and will change as this mix changes. We now discuss the implications of these uncertain ILUC effects for the design of policy to induce the appropriate mix of biofuels.

6. POLICY IMPLICATIONS OF INDIRECT LAND USE CHANGES OWING TO BIOFUELS

The ILUC estimates discussed above, show significant variability across studies and even within a study (depending on assumptions). These differences stem from the need to rely on imperfectly measured parameters to describe the responsiveness of the behaviour of economic agents to price signals and the technological and biophysical environment within which they make decisions. Plevin et al. [6] find that the resulting ILUC factor for corn ethanol ranges between 10 and 340 g CO\textsubscript{2} per MJ but is definitely non-zero. The presence of an ILUC effect on the GHG impact of biofuels raises the policy-relevant question of how government policies should be designed to respond to it and which policies should take it into consideration. In particular, should the ILUC effect be incorporated in the design of all policies that encourage biofuel production, such as the RFS, LCFS and RED or also in the design of broader policy initiatives, such as an economy-wide cap-and-trade policy? We discuss these issues below.

Environmental economic theory shows that in a first-best setting with only one externality, the socially optimal way to internalize the externality is by pricing it at the level equal to its marginal social damage, and holding producers accountable for the externalities they generate. Applying this to the problem of mitigating global warming requires a policy intervention that leads to a price of carbon or a carbon tax set equal to its marginal social damage, with the latter being determined by the environmental costs of global warming. Under a cap-and-trade policy, the price of carbon would be equal to the price at which the demand for tradable permits is equal to their supply and the cap is met. Performance-based policies such as a carbon price targeted directly at the externality are shown to be the socially efficient approach to mitigating the GHG externality compared with technology standards that mandate the use of particular technologies [46].

Environmental economic theory also distinguishes between two types of externalities, technological externalities and pecuniary externalities. The former are those where the actions of an individual directly affect the well-being of others, while pecuniary externalities are those where the collective actions of a group of individuals affect market prices which in turn affect the well-being of others. With a technological externality, like pollution, there is a divergence between social costs (the well-being of all) and private costs. Free market equilibrium will lead to over-production of pollution as private costs are lower than social costs. Government intervention is needed to price pollution or impose quantity limits to remove this wedge between private and social costs. In contrast, pecuniary externalities do not drive a wedge between private and social marginal costs. For example, a firm that increases its workforce in a small town could raise the price of labour and impose a pecuniary externality (in the form of lower profits) on other firms. While this reduces the benefits of other employers in the town, these are fully offset by the gains to the workforce in the town. Thus, pecuniary externalities do not affect net social benefits and no state intervention is required to ‘correct’ for these pecuniary externalities, if employers and workers have equal weight in net social benefits. While pecuniary externalities indicate a well-functioning market where prices respond to re-establish equilibrium when aggregate demand or supply for a commodity changes, technological externalities are a sign of market failure; they arise because there is a missing market (for pollution) and hence no price is paid for pollution by polluters. A technological externality therefore requires government intervention to impose a price on pollution or set a cap on the quantity of pollution. In a first-best setting, all polluters should be penalized for the technological externalities they generate. With a global externality such as GHG emissions, this would imply that producers all over the world should be penalized for any GHG emissions generated, including those owing to changes in their land use decisions that could cause carbon stored in soils and vegetation to be released. The analysis by Wise et al. [47] and Mellilo et al. [48] shows the importance of pricing emissions not only from fossil carbon but also from biogenic carbon for reducing ILUC. They show that taxing only the former will lead to large deforestation, with large conversion of natural ecosystems to either food or energy production. In contrast, when biogenic carbon is also taxed, forested land increases. Even with a global carbon policy that prices fossil carbon and biogenic carbon, there would still be ILUC but this would be a pure pecuniary externality that does not justify government intervention in and of itself because the ILUC-related emissions in this case would be at their optimal level.

Policy choices become less clear (from a theoretical perspective) in a second-best setting where (i) only a
subset of polluters can be penalized for their GHG emissions and/or (ii) the policies to control GHG emissions are not targeted directly at GHG emissions but are technology or intensity standards like the RFS/RED and LCFS, respectively. In the case of (i), the imposition of a carbon price in a few countries could lead to higher commodity prices in the world market, which would induce unpunished producers to increase production in a manner that raises their emissions. Although ILUC originates as a pecuniary externality caused by biofuel-producing countries, it creates a technological externality because of missing markets for carbon in other countries and affects GHG emissions that have global effects (and are not limited to the countries where ILUC occurs).

In (ii), the policies to control for GHG emission externalities in the biofuel-producing countries are technology/intensity standards and not the first-best carbon price policy. The motivations for these biofuel policies are mixed and include energy security, stimulating innovation that reduces dependence on exhaustible oil and rural economic development. While these mixed motives could be complementary, there may also be cases where they conflict. The presence of the ILUC effect of biofuels could be one such case because it shows that policies oriented to stimulating renewable energy/biofuels policies may not achieve a global reduction in GHG emissions, at least in the near term. Furthermore, these policies do not guarantee a reduction in GHG emissions even if there were no ILUC effects; the biofuel mandate lowers the price of petrol and while it raises the cost of biofuels, it could result in overall lower prices of the blended fuel and an increase in vehicle miles travelled. This could offset some of the petrol displacement benefits of biofuels. The net impact of the mandate on GHG emissions is theoretically ambiguous; it is shown to depend on a number of factors such as the ease of substitution of biofuels for liquid fossil fuels, the responsiveness of liquid fossil fuel prices to a reduction in demand and the responsiveness of vehicle miles travelled to the cost of fuel [49,50]. For similar reasons, Holland et al. [51] show that while an LCFS reduces GHG intensity of transportation fuel, it may not reduce GHG emissions. These policies may, therefore, require society to make trade-offs in the extent to which the multiple social goals of energy security, rural economic development and GHG mitigation are met through biofuels. In these second-best settings, a more careful cost–benefit analysis is needed to determine the appropriate policies to deal with leakage effects and the extent to which ILUC should be factored into policy design. The multiple motivations for biofuels together with the uncertainty about the ILUC effect suggest that we need to be more cautious in including an ILUC factor in the design of policies and that the cost of ILUC may be significantly overestimated because it disregards the contribution of biofuels to other social goals. Setting an ILUC factor too high could prevent innovation in the biofuel industry in the US and the EU. It could also be counter-productive if it enables biofuel production to move to other regions where even its direct effects on the environment may be much more detrimental.

There is an additional complication with including an ILUC factor in policies such as an LCFS or a national cap-and-trade policy. This arises because the ILUC-related GHG intensity is determined simultaneously with the GHG intensity used to implement these policies, as the latter determines the mix and quantity of biofuels consumed. This is unlike the case of the RFS where the ILUC effect can be simulated ex-post (with the implementation of the mandate) and it is possible to examine if specific biofuels do or do not meet the GHG intensity reduction requirements with the inclusion of the ILUC effect. In the case of an LCFS, the GHG intensity of each type of fuel needs to be specified ex-ante to policy implementation and will influence the mix of fuels used to meet the intensity standard and thus the ILUC effect of each biofuel. Including an exogenously specified ILUC factor ex-ante for a specific biofuel would be incorrect as it is unlikely to match the ex-post ILUC effect that occurs after the policy is implemented.

Furthermore, the presence of these market-mediated effects raises another issue; which indirect effects should be included in policy design, as ILUCs are not the only market-mediated effect of biofuel production. By displacing liquid fossil fuels, large-scale production of biofuels is expected to lower fossil fuel prices; this would offset at least some of the reduction in fossil fuel caused by biofuels. Additionally, it could also create incentives for fossil fuel producers to react to lower expected prices of fossil fuels in the future and increase extraction volumes to convert the proceeds into investments in capital markets that offer higher yields [52].

Moreover, existing policies like the RFS and the LCFS in the US do not prevent the increasing global use of oil sands. These are more GHG-intensive than conventional crude oil.7 The RFS does not limit the use of these sources domestically in the US and while a national LCFS would restrict their domestic consumption and imports into the US, it would not prevent the diversion of Canadian oil sands to other countries such as China; thus leakages of GHG emissions are not limited to those owing to ILUC. There are many indirect effects of biofuel production that arise owing to leakages and would need to be evaluated to determine which ones are significant enough to be incorporated. Accounting for only some market-driven indirect effects would be arbitrary and accounting for all of them would make policy design even more complex and cumbersome than it already is.

Furthermore, even with a national carbon policy, adding ILUC to the GHG intensity of biofuels would require adding it to the GHG intensity of all products.

7Oil sands, also known as bituminous sands or tar sands are a type of unconventional petroleum deposits that contain naturally occurring mixtures of sand, clay, water, and a dense and extremely viscous form of petroleum technically referred to as bitumen. Liquid fuel from oil sands is found to be up to 21 per cent more GHG-intensive than fuel from conventional crude oil (Jacobs Consultancy, 2009. See http://www.albertainnovates.ca/media/15753 life%20cycle%20analysis%20of%20oil%20sands%20biofuels%202007%20report.pdf).

US imports of crude oil from oil sands (mostly supplied by Canada) are increasing and could reach 20–30% of US oil imports in 2030 (IHS CERA, 2010. See http://www2.cera.com/Oil_Sands_SR_042110.pdf).
that cause ILUC. For example, a carbon price raises the costs of fertilizers and would reduce fertilizer applications. This could reduce crop yields as well as cultivation on marginal lands and lead to ILUC effects as world prices increase. Moreover, separating out the ILUC effects of increased fertilizer prices from those of increased biofuel production would be extremely difficult. Thus, if the policy incorporates the ILUC effects of biofuels it will need to include ILUC effects of all goods that affect market prices or make subjective decisions that are likely to be contentious.

As mentioned above, the assumed time horizon in the calculation for ILUC plays an important role in determining the per unit ILUC factor for biofuels and the arbitrary choice of a time horizon makes policy design with ILUC subjective rather than based on certain science. As an alternative, Mellilo et al. [48] suggest attributing the entire GHG effect caused by ILUC in a given year to the biofuels produced in that year. Their estimates would imply that in the first year of biofuel expansion, the entire indirect emissions caused by biofuels in that year (300 g CO₂ per MJ) be added to the GHG intensity of biofuels when determining their carbon price. This would result in a GHG intensity of biofuels that is three times higher than that of petrol and impose a heavy upfront penalty on biofuels that would decline over time. Such a policy however, would make any biofuel production prohibitive given that the costs of producing biofuels are higher than those of petrol to start with. This would require biofuel producers to face large losses in the near term and would be virtually equivalent to banning biofuels.

Including an ILUC factor in the design of existing policies, such as the RFS or the LCFS does not imply that GHG emissions will be reduced to their optimal level. While it will cause a switch towards biofuels with smaller ILUC factors, it does not ensure that global GHG emissions will be reduced and that social welfare in the biofuel-producing countries will be increased, especially given multiple objectives for their biofuel policies. It is also important to note that even without ILUC, these policies are not guaranteed to reduce GHG emissions or increase welfare. Whether the second-best policy for GHG mitigation should include a zero ILUC effect or the entire ILUC effect or something in between is an empirical matter. The design of these second-best policies for GHG mitigation should consider the extent to which the addition of the ILUC factor in the implementation of the RFS or the LCFS reduces global GHG emissions and the economic costs of doing so. Given the other objectives of biofuel policies, these costs should include the lost benefits from delaying the innovations and learning needed to transition to a low-carbon economy, reducing the dependence on oil and achieving a long-term reduction in GHG emissions. They should also include any social and environmental costs of biofuels, including their impacts on food prices, biodiversity and water quality. From a practical perspective, valuing these costs and benefits is complicated.

Can biofuels be a solution to climate change even if its production leads to ILUC? The answer to this needs to be nuanced by the type of biofuel being considered, because the DLUC- and ILUC-related GHG intensity of biofuels differs across the feedstocks used. Moreover, biofuels are currently not viable in the absence of government intervention (with the exception of Brazil); thus, government policies to directly support biofuels (in the absence of a global carbon policy) play a critical role in determining whether or not biofuels can mitigate climate change. Firstly, these policies need to encourage biofuels from feedstocks that have a much larger potential to reduce GHG emissions (including DLUC and ILUCs) compared with liquid fossil fuels. Secondly, these policies need to be accompanied by land use policies in the biofuel-producing countries that prevent conversion of natural vegetation and forests to energy crops and practices which ensure that the soil carbon sequestered by these energy crops is maintained permanently. Thirdly, biofuel production needs to be accompanied by sustainability standards for fuel blenders in the US and the EU which would create incentives for them to ensure that domestic and imported biofuels are produced in environmentally sustainable ways. Fourth, sustainability standards and land use management policies are needed in regions where ILUC could occur to reduce incentives for conversion of natural lands to produce crops displaced by biofuels.

In addition to appropriate government intervention, market-based pressures for sustainable land use practices could also play a role in making biofuels a solution for climate change mitigation. There are numerous examples of such standards leading to public–private partnerships between firms, government and non-governmental organizations to achieve a common goal to encourage a shift towards sustainable forest products and voluntarily reduce toxic releases and GHG emissions [53]. The effectiveness of such arrangements in environmental protection depends among other things on the pressure from market participants (consumers, investors and environmental interest groups) for environmentally friendly products, with the presence of systems that make environmental performance more transparent and hence easier to hold entities accountable, monitoring and reporting of environmental outcomes, and a credible threat of regulations to deal with violations [54]. There is some evidence that growing global demand for tropical food, timber and biofuels, and demand for environmentally friendly products by Western consumers as well as pressure by environmental interest groups is making firms that operate in tropical countries and sell to Western markets much more responsive. The effectiveness of these market-driven pressures for products produced using sustainable methods of production depends on how widespread they are, on consumer willingness to pay premium prices for such products, and the credibility of sustainability certification standards. The enforcement of these standards requires international agreements among countries. They also need to be supplemented by support from governments in countries where ILUC is likely to occur. This support could take the form of carrots and sticks, including a willingness to enforce laws and to provide payments for ecosystem services generated by landowners [55].
These could include direct payments to landowners to maintain forests and premium prices for commodities produced without clear-cutting forests. Such efforts have been initiated in Brazil, but determining the appropriate level of payments to provide appropriate incentives remains a challenge [56]. These efforts also need to be supported by international agreements that enable collective action among countries and is unlikely to be achieved by an individual biofuel-producing country or state (such as California) that seeks to remedy its own ILUC effect.

7. CONCLUSIONS

The potential for ILUC due to biofuel expansion and the possibility that it could more than offset the GHG savings from liquid fossil fuel displacement in the short run has led to concerns about policies that support biofuels. In a first-best setting, the optimal approach to deal with a global externality such as GHG emissions is to have a global carbon tax or cap-and-trade policy. In the absence of a global carbon policy, any unilateral policy to deal with GHG emissions, for example through a national carbon tax or through promoting alternative fuels such as biofuels will have numerous leakage effects, such as ILUCs and GHG emissions that result from them. With a global carbon tax, ILUC is a pecuniary externality that does not justify government intervention; global emissions are at their optimal level. However, without a global carbon tax, there are missing markets for carbon in countries without a carbon policy; as a result ILUC-related emissions become a technological externality that should be internalized. Hence, a case could be made to include an ILUC factor in the GHG intensity measure of biofuels used for policy implementation. The inclusion of an ILUC factor will serve as a ‘pseudo-tax’ that lowers ILUC emissions. However, given the wide range of ILUC estimates obtained by recent studies, determining the ‘right’ ILUC factor to include in regulations is likely to be controversial. Given the modelling uncertainties in determining an ILUC factor owing to its dependence on a choice of model, on a particular set of assumptions (including time horizon) and a particular policy mix, any factor chosen to capture ILUC and more broadly, other market-mediated effects for policy implementation will be subjective. The existing literature reviewed above shows that the ILUC effect is positive but the exact magnitude is policy- and baseline-specific and that it depends on assumptions that often have little empirical support. Given the differences among modelling approaches, it is unlikely that estimates will converge to a single number. The assessment of the ILUC owing to biofuels will get even more complex as advanced biofuel pathways and other policies to support biofuels (such as tax credits, tariffs and R&D subsidies) are considered. Simulating ILUC effects while incorporating advanced biofuels require a multitude of additional assumptions about the evolution of technology for second-generation biofuels, the viable mix of biofuel feedstocks and the utilization of co-products resulting from advanced biofuel production. Moreover, if policy-makers attempt to control for one leakage they would need to control for other leakages as well, such as those owing to the rebound effect. Determining the magnitudes of those leakage factors and will only compound the uncertainties already encountered.

At best, economic models should be used to provide probability distributions of the ILUC factor that reflect the uncertainties inherent in the assumptions that underlie it as in Plevin et al. [6]. While this is useful to obtain an understanding of the GHG mitigation potential of biofuels, it may not be particularly useful for designing policies such as a carbon tax or an LCFS, which needs to be based on simple estimates of ILUC. Whether that simple estimate should be the mean, median or mode of these probability distributions needs to be guided by a cost–benefit analysis that considers the multiple economic and environmental objectives motivating biofuel production.

An alternative policy approach is to minimize ILUC by inducing a transition to those second-generation biofuels that have little or no ILUC effect, and also creating incentives for sustainable land management practices, such as land zoning regulations and payments to landowners for environmental protection. Additionally, market pressures from environmentally conscious consumers, regulatory pressures on blenders of biofuels and sustainability standards developed by non-government organizations could create the incentives to transition to a more sustainable bio-based economy.

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