REVIEW

Accounting for indirect land-use change in the life cycle assessment of biofuel supply chains

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The expansion of land used for crop production causes variable direct and indirect green-house gas emissions, and other economic, social and environmental effects. We analyse the use of life cycle analysis (LCA) for estimating the carbon intensity of biofuel production from indirect land-use change (ILUC). Two approaches are critiqued: direct, attributional life cycle analysis and consequential life cycle analysis (CLCA). A proposed hybrid 'combined model' of the two approaches for ILUC analysis relies on first defining the system boundary of the resulting full LCA. Choices are then made as to the modelling methodology (economic equilibrium or cause–effect), data inputs, land area analysis, carbon stock accounting and uncertainty analysis to be included. We conclude that CLCA is applicable for estimating the historic emissions from ILUC, although improve-ments to the hybrid approach proposed, coupled with regular updating, are required, and uncertainly values must be adequately represented; however, the scope and the depth of the expansion of the system boundaries required for CLCA remain controversial. In addition, robust prediction, monitoring and accounting frameworks for the dynamic and highly uncertain nature of future crop yields and the effectiveness of policies to reduce deforestation and encourage afforestation remain elusive. Finally, establishing compatible and comparable accounting frameworks for ILUC between the USA, the European Union, South East Asia, Africa, Brazil and other major biofuel trading blocs is urgently needed if substantial distortions between these markets, which would reduce its application in policy outcomes, are to be avoided.

Keywords: consequential life cycle analysis; indirect land-use change;
carbon intensity

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1. INTRODUCTION

1.1. Background

Land-use change (LUC) from biofuels, and in fact any form of new demand on land and its products, can induce several economic, social and environmental effects. Direct greenhouse gas (GHG) emissions have been shown to be associated with land conversion from its 'original' state (forest, grassland, pasture, cropland, degraded land, etc.) to an altered 'state' that results from the production of biofuel feedstocks. Indirect land use change (ILUC) results in displacement effects, including price-induced changes in global commodity markets, that, in turn, also lead to land being altered from one state to another, with resulting changes in GHG emissions and carbon stocks on that land. Estimating an overall net ILUC GHG emissions value for a specific biofuel involves complex modelling. A coupled modelling framework is needed to estimate the impacts of the conversion of land between ecosystem types and the resulting balance of carbon stocks over time, with associated storage or release of carbon and other GHG species [1−9].

Life cycle analysis (LCA) has been used for decades to model system pollution and resource flows directly attributed to the producer and relative to a functional product unit. Life cycle inventory (LCI) is identified, and collated into the building blocks of inputs and outputs, then translated into indicators about the product systems' potential impacts on the environment, on human health, and on the availability of natural resources [10,11]. LCA has evolved in transportation fuel analysis to measure the energy and emission impacts of advanced vehicle technologies and new transportation fuels; the fuel cycle from wells to wheels (WTWs) and the vehicle cycle through material recovery and vehicle disposal [12]. Current work in biofuels sustainability evaluation focuses on the GHG emissions compared with a fossil fuel baseline from LCA estimates, and is divided between the use of the attributional versus consequential (ALCA and CLCA) approaches.

The objective of the ILUC modelling approach is to estimate climate-change impacts arising from changes to the net release of GHGs that, in turn, result from the substitution of one fuel for another. By comparing one megajoule (MJ) of fuel with another, the result is the difference in the *physical global warming intensity* (GWI_p) values of the two fuels. The GWI_p for each fuel is the sum of its direct and indirect emissions measured as grams $CO_2e\,MJ^{-1}$. In the following pages, we assess the GWI of a fuel in the sense of it being a measure of carbon intensity (CI) notionally applicable to evaluating the climate effect of this physical substitution.[1] Multiple accounting systems or metrics are required to measure (or estimate) specific components of the GHG emissions of supply chains that can originate on different land types in different regions. These supply chains also result in different products, but affect the same markets.

The challenge in estimating land-use change effects of biofuel expansion include a number of different modelling and biofuel scenario projection issues, such as:

— data issues;
— carbon accounting;
— multiple indirect effects;
— time treatment; and
— uncertainty in a wide range of factors.

Herein, the key assumptions, models employed, and interpretation of results are analysed.

The attributional life cycle analysis (ALCA) approach provides information about the direct emissions from the production, consumption and disposal of a product, but does not consider indirect effects arising from changes in the output of a product. ALCA generally provides information on the average unit of product and is useful for consumption-based carbon accounting. A further expansion of ALCA methodologies includes the PAS 2050,[2] in which stakeholder input into the LCA is specified as a life cycle assessment of the analysis: *specification for the assessment of the life cycle greenhouse gas emissions of goods and services*, and to a large extent ISO 14044 *Environmental Management—Life Cycle Assessment— Requirements and Guidelines*.[3] An ALCA informs comparisons between the direct impacts of products, and is used to identify opportunities for reducing direct impacts in different parts of the life cycle. The allocation method is most often used in this approach to account for the impacts of co-products.

Consequential life cycle analysis (CLCA) aims to provide information about the net change in system emissions caused by a change in the level of production of a product. CLCA is useful in trying to understand the total GHG consequences from changing the level of production for a product, and is therefore most appropriate for policy appraisal. Co-products are treated by system expansion in CLCA and are evaluated on a similar spatial and temporal scale as biofuel production [14,15].

ALCA, therefore, measures environmental flows from and to a system and its subsystems, while, in the CLCA method, these relevant flows change in response to economic signals transmitted through the world economy often far, both physically and causally, from activities directly associated with fuel use [16]. CLCA is highly dependent on projections of the future, and understanding of the past, and requires what-if scenarios and proposed counterfactual circumstances. In contrast to ALCA, the system boundary in CLCA expands through consequential runs, to estimate marginal products affected by a change in the physical flows in the central life cycle. CLCA is currently the

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[1]This measure is a step along the way to, but not the same as, an *administrative* CI_a that should be assigned to the fuel in a policy context. This difference is discussed in Hare *et al.* [13] and not further considered here.

[2]PAS 2050 is a publicly available specification that provides a method for assessing the life cycle GHG emissions of goods and services (jointly referred to as 'products'); developed by the BSI Group (a global business services organization providing standards-based solutions in more than 150 countries).
[3]ISO 14040:2006 describes the principles and framework for LCA, including: definition of the goal and scope of the LCA, the LCI analysis phase, the LCI assessment phase, the life cycle interpretation phase, reporting and critical review of the LCA, limitations of the LCA, the relationship between the LCA phases, and conditions for use of value choices and optional elements.

model approach 'choice' by regulators, and academics to estimate effects such as ILUC and global market effects from biofuel production.

1.2. *Defining the system boundary and life cycle inventory*

Several parameters are involved in averaging data inputs. The initial 'choice' in LCA is how to delimit the system boundary and this choice will ultimately affect what to estimate in the upstream versus downstream of the biofuel production process, and further to indirect land 'use' (e.g. elsewhere), or global market effects. All LCAs for fuels confine the boundary of the fuel production pathway to a manageable system. Figure 1 identifies the system boundary for the 'general' biofuel production pathway.

The CLCA method requires choices that are often not transparent in analyses. Table 1 separates these choices and compares the data parameters. They include: the definition of the 'system', the treatment of co-products, carbon (GHG) emission factors (EFs; inclusive of farming practice), data uncertainty (including in the values used to assess each parameter), world market flux, predictions about the future trends in production technology (including yield improvements) and estimates of historical and changing land use and the carbon stocks of land types used to estimate EFs. While several other issues are important, this paper evaluates only land-use change and how GHG emissions are calculated by LCA approaches for estimating direct and indirect GHG emissions from LUC from biofuels [17–19], and arising from new demand for biofuels. ALCA and CLCA model the same process quite differently and the key difference between ALCA and CLCA is the choice of boundary and whether the dynamics of a system are considered or not.

The key drivers for choosing ALCA involves simplicity and the availability of average, as opposed to the marginal, data needed for CLCA. ALCA analyses include all the emissions under direct control of the sequence of production process operators, at the production sites (farm, biorefinery, etc.), whereas the CLCA includes all effects whether under the control of the operator or not (such as all significant indirect contributions that change global GHG concentrations). This leads to the argument that perhaps LCA cannot provide what it is ultimately meant to calculate: the total impact of a production system (or policy); followed by the question addressed in this review: does CLCA represent a more accurate sum of the consequences of the perturbation of a system or of a policy? Our conclusion is that ALCA and CLCA are complementary, because both perform different functions in the assessment of real production systems, yet the indirect effects from biofuels are difficult to model, estimate and therefore project.

2. ATTRIBUTIONAL AND CONSEQUENTIAL LIFE CYCLE ANALYSES MODELS AND LINKAGES

Several authors have reviewed the utility of LCA methods and models to analyse indirect effects, including future temporal scenarios and impacts for biofuels

outside the immediate production chain [1,4,13,20–24] (M. O'Hare 2010, unpublished data). Methods vary for estimating land conversion and associated GHG impacts to combine agro-economic model output with spatial and temporal data that estimate associated changes to the carbon cycle. The 'combined' model approach in CLCA involves econometrics models, although not all are publically available. These models can be separated by geographical scope, treatment of time, partial or general equilibrium, the type of analysis and estimates of GHG emissions from land conversion.

Models are based on different assumptions, internal structures, datasets, emissions and criteria pollutants tracked, and limitations on the different fuel pathways. For example, the Greenhouse Gases, Regulated Emissions and Energy Use in Transportation (GREET) model developed by Wang *et al.* at Argonne National Laboratories includes more than 100 fuel production pathways from various energy feedstocks and is designed with stochastic simulations to model uncertainty. This model has been adapted to specific regional fuel mixes, and mandates, such as the low carbon fuel standard (LCFS) in California to estimate direct emissions [25].

ILUC requires an expansion of the nested variables found in standard econometric models such as the Global Trade Analysis Project (GTAP). GTAP has, therefore, evolved from a classic econometric model and transformed for biofuel modelling from a full bilateral trade between world regions to accommodate biofuel ILUC estimates for those regions. For example, GTAP-AEZ (Food and Agriculture Organization (FAO)/ International Institute for Applied Systems Analysis (IIASA)), involves production with intra- and inter-regional land heterogeneity, represented by agro-ecological zoning (AEZs), that categorizes 18 different types of land within each region based on land characteristics (soil type, rainfall, etc.). GHG emissions and sequestration modifications for non-CO_2 and different classifications of emissions incorporate new detailed forest carbon stock data, and modelling of intensive and extensive carbon management options are just some of the additional variables considered in GTAP-AEZ (figure 2). The goal of this approach is to calibrate mitigation responses to partial equilibrium (PE) model responses, although there are limitations to this approach.

The measurement of overall CI for a biofuel pathway is often estimated as a single value, although it combines both direct and indirect effects. The California Air Resources Board (CARB) designed the LCFS to be based on the overall CI value of the fuel; with the intention of incentivizing improved fuel pathways with the lowest CI. CARB provides separate CI values for each feedstock (referred to as an ILUC 'risk adder'), derived from the combined econometric model approach, including sequential runs of GTAP. GREET has been adapted further for the LCFS, e.g. 'CA-GREET', to model specific biofuel pathways [27,28] for the direct emission estimates. The results are published as biofuel pathways on the CARB website. The US Environmental Protection Agency (EPA) uses a different approach for the renewable fuel standard (RFS2), where direct and indirect CI
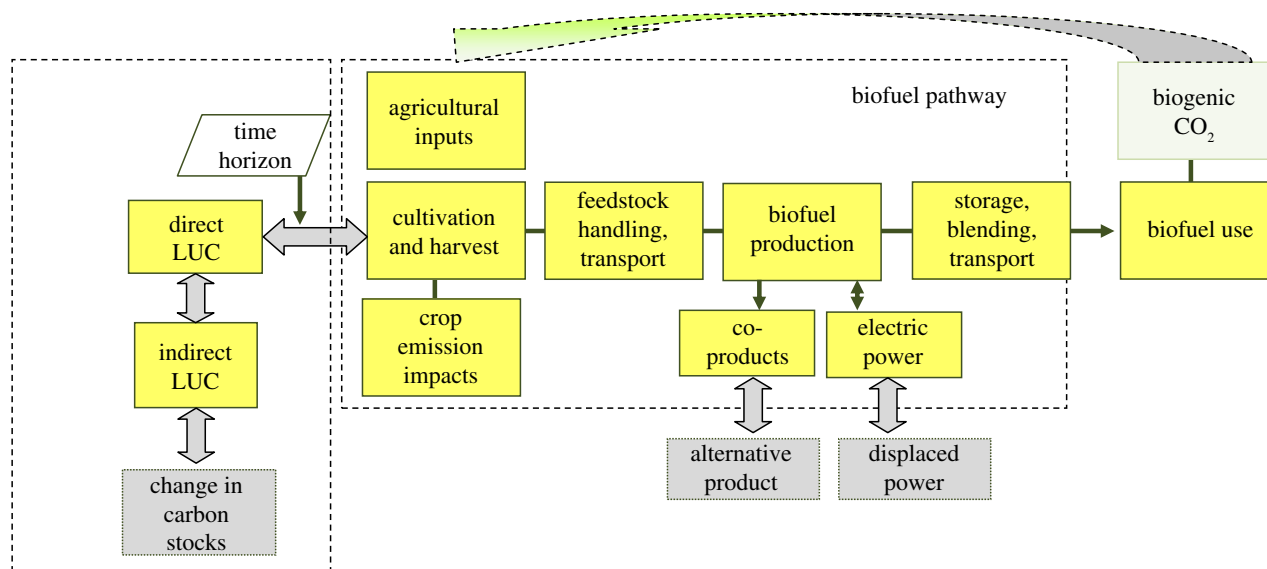
Figure 1. System boundary for LCA inclusive of indirect effects.

Table 1. Comparison of attributional and consequential analysis.

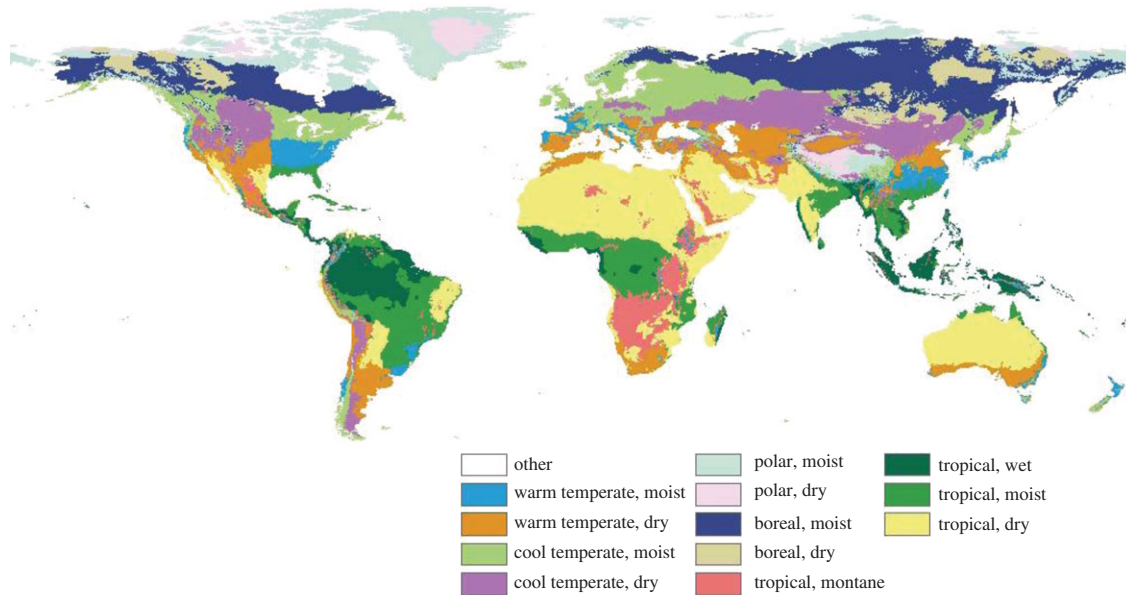| parameter	questions asked/attributional LCA	agricultural data/consequential LCA
questions asked	what is the global warming potential measured by the carbon intensity (CI) produced for an average unit of product? what is the CI for a specific fuel pathway?	what is the consequential change in total emissions as a result of a marginal change in production?
approach	calculate total direct (including direct+upstream) emissions from inputs and LCI vectors	model emissions associated with economic response to output and price effects
data	producer data inputs; using average data or default values	marginal data inputs price elasticities product demand and supply curves plus ALCA data
application of results	determine emissions associated with production of a specific product determine consumption-based emissions	inform policy-maker or consumer of total emissions and indirect effects (as much as possible) for a purchasing or policy decision
system boundary	system flows under direct or indirect control of the operator boundary may be expanded to capture important local effects	process flows within system boundary and outside of boundary indirect effects include market, constrained resource use, substitution effect; ideally all consequences
treatment of co-products	allocation or substitution method	substitution with second-order or indirect substitution effects including market-mediated effects
agricultural data	average or marginal data	historical and projections; Food and Agriculture Organization of the United Nations statistical service; Food and Agricultural Policy Research Institute; other outlook models
model approach	spreadsheet or database models with interlinked pathways and circular references	general equilibrium (GE) LCA flows; partial equilibrium (PE) (rebound effects); dynamic (improve understanding of marginal system effects). Separate, or combined with ALCA approach
market effects counted?	no (or with exogenous displacement factor)	yes
non-market indirect effects	generally no	depends on approach

[a]Adapted from Tipper *et al.* [18].

Figure 2. Agro-ecological zones (AEZs) of the world used to represent biophysical heterogeneity in the GTAP-AEZ model approach [26].

values are combined to provide an overall value, and therefore the indirect CI value is proprietary to EPA. Results are published as one value so you cannot distinguish an ILUC 'value' separately. One, or the other, combination is one of the options being considered by the European Commission in the Renewable Energy Directive to measure ILUC as a CI value, or 'score' per biofuel pathway.

The double-counting approach is seemingly justified by categorizing emissions into 'direct' and 'indirect' categories and using allocation for 'direct' emissions and substitution or system expansion for 'indirect' emissions. However, this is better characterized as the root of the confusion rather than a methodological justification.

2.1. Sources of uncertainty related to indirect land-use change

ALCA provides an inventory of the emissions directly associated with the lifecycle of a product—but not the total system change in GHG emissions caused by a change in the production of the product, i.e. it does not estimate the total impact of the policy, which is where CLCA has its application. Several sources of uncertainty have been identified in LCA modelling related to this, although the primary debate revolves around the methodological approach of using ALCA and CLCA to model ILUC.

While the original focus of ILUC analysis was centred on corn (maize) ethanol production most of the world's current feedstocks for biofuel production have now been assessed using a wide range of modelling techniques (economic and non-economic) including sugarcane ethanol [6,29,30], wheat (e.g. [6,7,31]), palm oil [6,7], etc. The model approaches discussed previously continue to expand to include evaluation of specific pathways, in different regions of the world, from very different feedstocks.

2.1.1. Data issues
This section compares and contrasts the issues of data requirements, availability and uncertainty for ALCA and CLCA. Table 2 summarizes our findings.

Several areas are identified in resulting CI values when disaggregated. For example, working backwards in the EPA analysis for RFS2, significant uncertainties are clearly identified (e.g. the levels of soil N_2O emissions resulting from nitrogen fertilizer use, changes to carbon stocks of soils, how to account for co-products). EPA ran the RFS2 analysis through to 2022, using baseline data on carbon stocks for a 4 year period (this was later revised to 6 years), assuming a wide range of crop yields and technology improvements. EPA's analysis model approach includes all major emission changes, including land-use change and non-LUC emissions. Data required for CLCA include many sources that are at present less well understood and less well documented, although EFs for other indirect effects are beginning to be recognized as important for inclusion into overall CI values. Ongoing work estimating livestock emissions, rice cultivation, crop switching and differences in on-farm energy and agri-chemical use are important not only to expand our understanding of the range of emissions but also to identify uncertainty and gaps for further work.

While it is likely that the quality of data sources required for CLCA will improve in the near future, it is critical to estimate uncertainty, particularly in a modelling approach that includes several default parameters and scaled-up historical datasets.

2.1.2. Hybrid indirect land-use change model approaches
Combining ALCA and CLCA may have practical advantages, i.e. ALCA is easier for reporting fuel suppliers, but it can be methodologically confused and, when it is, it produces neither truly 'direct' nor 'indirect' results. When combined with CLCA, this lack of

Table 2. Comparison of data requirements, availability and uncertainty for fuels ALCA and CLCA*.

ALCA	CLCA
data requirements	in addition to the data required to perform the ALCA, CLCA inputs include:
1. data specific to the *activity* within the LCA boundary (e.g. amount of fuel produced, amounts of material and energy inputs consumed in the farming, production and refining processes, transport of fuels, amounts of waste generated)	1. market impact data relating to increased biofuels production (e.g. price elasticity data for biofuel inputs and outputs, response of world markets to increased production of biofuels and biofuel co-products)
2. data on the *environmental impacts* of manufacture and supply of finished fuels and energy inputs	2. technical response data to increased biofuel production (e.g. crop yield response to increase in demand and price of biofuel inputs, crop technology potential, impact of increased farming inputs, impact and potential of modified farming practices, potential and impact of disused, abandoned, marginal and degraded land, impact of biofuel-driven sustainability criteria on farming outputs)
data availability	
type 1 inputs: theoretically readily available data which can be monitored over time, to create average values for input to the LCA. The actual availability, quality and accuracy of these data depend on three key factors:	type 1 inputs: many potential sources (e.g. US Department of Agriculture or Organization for Economic Cooperation and Development). However, all have yet to fully factor in the effects of the imminent expansion of the biofuels market; not only in terms of increased demand for agricultural products for biofuel feedstocks, but also in terms of the effects on some agricultural markets of the large quantities of biofuel co-products which will be entering the animal feed markets (e.g. distiller's dried grains with solubles)
— accuracy of the measurements made (e.g. accuracy of weighbridge used to measure the amount of wheat entering an ethanol mill)	
— rigorousness of the monitoring process, which may be supplemented with a verification process	
— ease of access to commercially sensitive information	
type 2 inputs:	type 2 inputs: the available data are based mainly upon historic trends and in most cases are dependent upon well-understood technical factors. However, because these data are likely to be derived from historic trends, the impacts of the emerging biofuels market itself on their future values cannot be factored in
— at present, there are many examples where producers have not yet carried out accurate LCAs for their products. In these cases, biofuel LCA practitioners are required to use 'average' or default values	
— if a biofuel producer identifies and selects an input with very low embodied environmental impacts (e.g. chooses his feedstock from a farmer that minimizes GHG emissions), hence removing that material from the market, causing other potential users of the same type of material to use more carbon-intensive suppliers. This creates an indirect impact in itself, and brings in the need for a CLCA approach	

(*Continued.*)

Table 2. (*Continued.*)

ALCA	CLCA
data uncertainty the greatest uncertainties in input data are where direct, accurate measurement has not yet been carried out, or is prohibitively expensive, or where values vary significantly over time. The most striking example is the uncertainty of nitrous oxide emissions from soils, which can vary dramatically from region to region and even between adjacent fields; while default values are available, there is still considerable uncertainty in these default values [32]. Units used for reporting quantities of fuels and materials consumed may vary among industries and regions, challenging the reporting and documentation in fuel LCA models. Nonetheless, like quantities should be reported in a consistent manner (perhaps repeated in metric units)	type 1 inputs: — projected supply and demand data: data influenced by multiple factors, many of which are interdependent and may be influenced by biofuel expansion itself — price elasticity data: based on historic trends. Any structural change in world trade could induce changes that cannot be reflected from the historic data type 2 inputs: — crop yield response to increase in demand and price: three factors affect this input: — crop technology potential: new techniques in breeding and selection are constantly emerging. The rate of development and potential of new, as yet unknown, technologies is impossible to predict accurately — increased farming inputs: difficult to forecast and highly variable. Fertilizer use is particularly difficult to project where very different application rates and methods are common within and between farms in the same region and the data would also be sourced from economic models which cannot reflect real-time farming practice — farm practices: the likely impact of improved farming practices is also uncertain and difficult to model. The main area of uncertainty is the interrelationship between market factors and the implementation of the improved practices — potential of disused, abandoned, marginal and degraded land: significant potential for cropland expansion into this type of land [33,34]. Although most models do not yet factor in the potential impact of biofuels expansion on this type of land, any impact will be strongly policy-driven. The inevitable uncertainty about the extent and the direction of government policies in different world regions is certain therefore to add significant uncertainty to modelling this parameter. (See above for more detail) — impact of biofuel-driven sustainability criteria on farming outputs: European biofuel suppliers must comply with strict sustainability criteria. There are as yet no data available on the likely impacts of this requirement and it is not factored into existing modelling capability

methodological clarity of the ALCA can double-count the 'benefit' of co-products. The problem arises if the ALCA for the direct emissions allocates emissions to co-products, and the estimation of ILUC includes a credit or reduced net ILUC figure owing to co-product substitution effects, as the benefit of the co-products will be counted twice. The result could over-estimate the GHG savings from biofuels.

If the total GHG consequences arising from biofuel production (total system change in emissions per additional unit of biofuel produced) need to be estimated, then a CLCA should be undertaken. A pure CLCA treats co-products only once using a substitution or 'system expansion'.

2.1.3. Consequential life cycle analysis: accounting for multiple indirect green house gas emissions

The US EPA has recognized that multiple significant indirect GHG emission sources may be consequentially changed by biofuel production. In the LCA methodology employed in RFS2, the EPA attempts to quantify a multitude of indirect changes in the USA and global agricultural economies for several biofuel production pathways, including changes in domestic and international GHG emissions from farm inputs, land-use change, rice methane and livestock. The EPA estimates changes in these sectors using the Forestry and Agricultural Sector Organization Model (FASOM) and Food and Agricultural Policy Research Institute (FAPRI) economic models. The GREET model, along with EFs from the Intergovernmental Panel on Climate Change (IPCC), is then used to translate these agricultural and ecosystem changes into GHG emissions. By estimating indirect emissions in this manner, the EPA has tried to comply with US legislation, while recognizing the vast global complexity in various significant emission sources that are indirectly affected by biofuel production. In addition to the models above, the RFS2 methodology also uses data compiled from: CENTURY, DAYCENT, MOVES, FORCARB, NEMS and ASPEN[4] to project GHG emissions as a consequence of biofuel production. In total, at least eight highly complex models are employed to quantify direct and indirect GHG emissions from the corn ethanol life cycle.

In addition to land-use change, the aggregation of many other significant indirect emissions leads to a great expansion in analytical complexity. Because the EPA estimates *projected global* changes in GHG emissions from *all major* agricultural sectors and ecosystems for 15 years into the future, it is clear that such an approach that incorporates tens of thousands of parameters is likely to be associated with a large degree of error. We recognize that econometric models—dynamic, static or with different sectoral resolution—are by necessity 'uncertain' and that CLCA compares a baseline against a projected alternative scenario that aims to cancel out a large part of the uncertainty; yet, each projection associated with a sector change is associated with an uncertainty, and multiplying sector projections will add to total uncertainty.

[4]See EPA docket for all models and approach: http://www.epa.gov/otaq/fuels/renewablefuels/regulations.htm.

Table 3. Two alternative sets of estimates of the impact of three indirect emissions in CLCA.

selected indirect effects	EPA RFS2 [38] $gCO_2e\ MJ^{-1}$	other estimates $gCO_2e\ MJ^{-1}$
global indirect land-use change emissions (ILUC)	+30.1	+13.9[a]
global livestock	−0.28	−47.5[b]
US military emissions resulting from securing Middle East oil	0	−17.5[c]
sum of indirect emissions	+29.8	−51.1

[a]Adapted from Tyner *et al.* [36].
[b]Adapted from Liska & Perrin [4] based on Searchinger *et al.* [1] and Steinfeld *et al.* [39].
[c]Adapted from Liska & Perrin [40]: consequential reduction in US military in the Middle East based on LCA of US military and attribution of 20% (approx. $100B yr^{-1}) to oil security.

Consider the uncertainty in estimating indirect GHG emissions from corn production in the USA. Using reasonable parameter values, these estimates have ranged from 118 $gCO_2e\ MJ^{-1}$ [35] to 13.9 $gCO_2e\ MJ^{-1}$ [36], and have even ranged from 18.3 to 80.4 $gCO_2e\ MJ^{-1}$ within a single study based on uncertainty in economic projections [37]. In the EPA analysis, 30000 EFs are used to estimate emissions from land conversion alone. These EFs are one of two datasets included in the EPA's partial error analysis, leading to a 95% confidence interval that is ±28% of a mean value of 30.1 $gCO_2e\ MJ^{-1}$. The Searchinger *et al.* [1] model contained no specific land supply structure for various countries, and models with plausible land conversion supply curves appropriate for each country have not yet been published.

The issue is where to set the boundaries for CLCA, in particular with respect to the large number of indirect effects that can be considered in a system. Yet, it is clear that reasonable indirect effects that have similar magnitudes must be accounted for in parallel in CLCA. The difficulty in deciding where to set these boundaries, and the inadequacy of accounting for ILUC alone, can be further illustrated by assessing the cumulative impact of three uncertain indirect effects included in the RFS2 analysis when comparing US corn ethanol with gasoline produced from Middle Eastern oil.

— A recent estimate of ILUC emissions by Tyner *et al.* [36] provides a value of less than half of the estimate used by the EPA (table 3).
— For livestock impacts, analysis suggests that livestock populations may decline more owing to biofuels and livestock GHG emissions may be higher per unit, which together lead to greater indirect GHG savings than the EPA's estimate [4].
— Although ILUC associated with the extraction and production of fossil fuels was discussed by the EPA, some have argued that other indirect effects

associated with fossil fuels should be included. For example, GHG emissions, primarily arising from ship and plane movements, associated with the acquisition and defence of foreign oil for the USA [40,41].

The net cumulative impact of these three indirect effects within the full fuel cycle could amount to GHG emissions savings from corn ethanol as large as $51 \, gCO_2e \, MJ^{-1}$, which is more than $80 \, gCO_2e \, MJ^{-1}$ lower than the positive indirect GHG emissions that the EPA currently ascribes to US corn ethanol supply $(29.8 \, gCO_2e \, MJ^{-1})$.

This highlights the large degree of uncertainty (both in magnitude and direction) that exists in assessing multiple complex indirect effects simultaneously in CLCA, and the fact that one indirect effect may be offset by a series of other indirect effects. A recent report by Ensus [42] also indicates that there appears to be a structural bias in the way that equilibrium (economic) models have been used leading to a tendency to over-estimate the scale of indirect emissions from certain biofuel supply chains. These chains include wheat to ethanol, rape (canola) to biodiesel and corn (maize) to ethanol, i.e. those that produce protein-rich co-products that can be used as animal feed.

If robust estimates of the overall net GHG emissions from biofuel supply chains are required then it is likely that the boundaries used in CLCA will need to be expanded beyond quantifying ILUC emissions to encompass other significant drivers of indirect emissions. Currently, it remains unclear how to define these boundaries consistently, but it is clear that effects of similar magnitude should be analysed.

2.2. Other impacts: food

The consequences of significant land-use conversion to biofuel crops may have major implications for food security, biodiversity and soil and water quality. The displacement of existing land-use for biofuel production (biofuel crop area expansion) increases the pressure on other types of land use [43]. A key variable is the diet, especially the shares of meat and dairy, which exert a large leverage on land use owing to pasture and feed. Developing the capability to quantify these impacts and to include them in an LCA remains a major challenge. How the displaced activities, such as food production, are relocated will establish the magnitude of the impact of ILUC, which will be determined by the availability of agricultural and uncultivated (e.g. set-aside, fallow and forests) land [22]. Recent modelling studies of climate change impacts on global food production and undernourishment [44] has supported FAO predictions that food production will have to increase by 70 per cent over the next 40 years to feed the world's growing population [45]. The FAO further stated that, with the world's population expected to increase from the current 6.7 billion to 9.1 billion by mid-century, if more land is not brought into use for food production now, 370 million people could be facing famine by 2050.

The Bioenergy and Food Security (BEFS) [46] project focuses on the management of the sector and provides an example of how case studies such as those run by BEFS in Peru, Tanzania and Thailand can also be integrated into a country's food security monitoring system (FAO). The BEFS framework helps us to understand the very complex issue of the linkages between bioenergy and food security and is intended to be diagnostic rather than prescriptive. It includes natural and social diagnostic analyses, and economic assessment, and intends to act as a policy tool.

Clearly, expanded biofuel production will have variable positive or negative impacts on food production, and estimated impacts are contingent on regional policies, decisions and assumptions made both as external to and within the system boundary of the analysis. Biofuels compete for land and resources needed for food production or, alternatively, they help to provide the investment in infrastructure and the energy inputs needed to enhance the productivity of food cropping, harvesting, processing and delivery. Approaches such as BEFS could augment existing policy frameworks by providing working examples of the complexities of bioenergy and their linkages, and develop frameworks that are applicable in the regional or localized context.

2.2.1. Carbon accounting and time treatment

EFs for land conversion are calculated based on carbon stock estimates and application of carbon stock accounting methods, specifically the IPCC Agriculture Forestry and Other Land Use (AFOLU) methodology [47]. High uncertainty in above- and below-ground carbon stock estimates is well known although the inclusion of this uncertainty in overall EFs is variable in application by different US policies for ILUC [37,48–50]. There are two central issues: (i) how to estimate the type of land cover and area estimated to change and (ii) how to combine several separate EF estimates for each land type, carbon pool and applied stock change factor, i.e. conversion, reversion and associated management factors.

Currently, the existing estimates of ILUC have used different carbon stocking and flux estimates and methods to calculate EFs. In the Searchinger *et al.* [1] analysis, historical carbon stock estimates from the 1990s were provided by Houghton & Hackler [48] using Woods Hole Research Center (WHRC) datasets for above- and below-ground biomass stocks separated into 10 regions of the world. EFs were based on IPCC (tier 1) methods and then combined with GTAP, to provide estimated areas and locations for the land predicted to be converted to farmland as a result of the increased commodity prices resulting from expanded demand for feedstocks for biofuels. To formulate GHG EFs for different conversion types (e.g. forest to cropland), these values include various assumptions about the land's prior vegetation type and the release period post-conversion. The EPA's RFS2 EFs were estimated by the Winrock team [50] and represent spatially explicit estimates for 314 key regions in 35 countries. The EPA applied EFs to the satellite-based remote sensing-mapped regions estimated to be converted owing to direct and indirect biofuel expansion in the future and conducted individual model runs for biofuel scenarios in the PE model FAPRI.

In addition to the potential one-off release of carbon,[5] the calculation should also account for any subsequent uptake of carbon to the soils as a result of the crop, e.g. as a result of carbon sequestered in root systems and in the above-ground biomass. Several publications report on these data, particularly for released carbon [51]. However, it is well documented that above- and below-ground carbon stocks are variable and, although model and data improvements are employed in more recent ILUC analyses [36], there is still a requirement to incorporate refined soil carbon data inclusive of variance in carbon density based on type, practice and measurements.

The updated EPA analysis for RFS2 includes several updated above-ground carbon estimates and now uses the Harmonized World Soil Database (HWSD) for below-ground carbon estimates. Another difference between the CARB and EPA EFs is the treatment of harvested wood products (HWPs), whereby Winrock did not include an HWP factor. The CARB analysis, however, does account for harvested wood in the sensitivity analysis by applying the IPCC default 90 per cent oxidized carbon (e.g. 10% retained in wood products). Finally, the uncertainty analysis by EPA included a Monte Carlo analysis, while CARB used weighted averages. The resulting EFs impact the overall CI differently and, while this topic is not fully explored here, CARB and EPA are evaluating updates to ILUC estimates for feedstocks inclusive of ILUC factors applied to the total CI.[6]

2.3. Treatment of biogenic carbon

The treatment of biogenic carbon is a complex issue that is closely tied to the treatment of LUC. Several metrics are possible for biogenic carbon and these are applied inconsistently among fuel LCA models. Figure 3 shows the concept graphically, comparing baseline fossil fuel with biofuel. Biogenic carbon can be treated as neutral, i.e. carbon emissions from combustion at the vehicle tailpipe and along the fuel supply chain are assumed to be balanced out by prior or re-growth of the biomass and its associated carbon fixation from the atmosphere.

Alternatively, all carbon emissions are counted along the full fuel supply pathway including end use and the uptake of carbon during the crop growth. In most cases, biogenic carbon is treated as neutral for biofuel crops but positive (i.e. as a net emission) for biogas arising from the anaerobic digestion of 'waste' materials. The treatment of biogenic carbon from waste materials requires further examination owing to the various
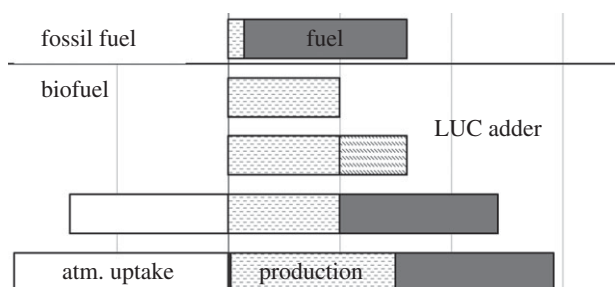

Figure 3. Schematic showing fossil fuel and biofuel cycle emissions with fuel carbon (detailing fuel combustion, downstream in black).

possible alternative fates of the carbon, e.g. re-use through recycling into a range of products with differing half-lives of the embedded carbon, disposal and long-term sequestration in landfills, disposal and rapid release as methane in landfills, disposal, rapid release and capture of methane and use as fuel for heat and electricity (see Brander *et al.* [21], for a comprehensive assessment of this issue). The quantitative amount of biogenic residues and wastes currently used for biofuels is small, though, and, in consequence, any inconsistency in accounting for C storage owing to the life time of biogenic materials used as products is comparatively small. Robust accounting will require consistently distinguishing between fuels whose use 'almost certainly' assures quick recapture of the carbon released when they are burned (e.g. annual crops and perennial grass feedstocks) and fuels whose carbon may or may not be recaptured. Time accounting for the carbon stocks and fluxes in longer rotation forestry and its multiple potential products is extremely complex and accepted methodologies for doing so are only just emerging [52].

As depicted in figure 3, emission sources may result from changes in biofuel production rates. To highlight the differences in LCA CI that can result from adopting different approaches to biogenic carbon, the LCFS GREET and Joint Research Centre (JRC) studies yield similar results for starch (corn and wheat)-based fermentation ethanol pathways as they use similar (neutral) assumptions for biogenic carbon. By contrast, the two studies produced strikingly similar emission results for waste forestry wood, but yield different results for farmed wood. The LCFS GREET calculates much lower GHG emissions than the JRC study because the biogenic carbon contained in the forest waste is deducted from the GHG emissions in the GREET model; the JRC study does not credit the biogenic carbon in the residues. JRC considers this to be consistent with the practice of not accounting for reductions in soil carbon by conventional cropping, and it has, at this time, estimated the effect on long-term forest C stock to be low.

2.4. The time horizon

The warming potential of biofuel GHGs requires a choice of treatment of the period of release. LUC-caused emissions typically occur at, or near, the beginning of the production cycle. The time horizon used to calculate global warming potential (GWP) differs in that the latter is used to estimate the useful life of

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[5]Searchinger *et al.* [1] assume all above-ground carbon is lost immediately upon conversion and 25 per cent is assumed stored in the ground. Further analyses [36] and the EPA (RFS2 2009) account for forgone emissions and differences in stored carbon. IPCC defaults are compiled estimates for soil and land-cover carbon profiles at a global scale with no specification of geographical distribution. New approaches in computable general equilibrium (CGE) modeling (GTAP-AEZ) use carbon data down to the AEZ level and the new harmonized global soil carbon database combined with FAO data on carbon-harvested wood stocks. For information, CGE models are a class of economic models that use actual economic data to estimate how an economy might react to changes in policy, technology or other external factors.
[6]Owing to decisions of the Cancun Conference in December 2010, developing countries will become subject to bi-annual GHG national reporting that takes into account LUC.

biofuel production infrastructure. The choices made when combining ILUC emissions with direct emissions usually requires a choice of amortization process; this is analogous to the combination of capital and variable costs of producing products. The GWP has been used in several analyses of LUC where the total indirect emissions from commencing fuel production are divided by the total fuel produced during a predicted production period, and this average is added to the direct emissions. This approach implicitly treats a unit GHG emission released today as though it has the same consequences as one released decades in the future [53]. Considerations affecting the allocation of ILUC to a unit of fuel include:

— *Amortization period* [54]. Some biofuels have much longer prospective production periods than others on grounds of cost (e.g. sugar cane ethanol versus corn ethanol; form factor/tractability) owing to different harvesting and ratooning (rotation) cycles. Also, non-biofuel alternatives may displace them (i.e. various electric vehicles). For example, assuming a 30 versus 100 year lifetime for a US corn ethanol production chain[7] results in substantially different CIs.
— *The use of discounting.* Although the employment of a 'discounting factor' to the time horizon is debatable, all GHG emissions for fuels being compared occur at the same time for each fuel, and may be regarded as reasonable proxies for warming (as of a given time, or accumulated over a long period, etc.). But if they are not, discharges need to be converted into warming (as a proxy for social cost) to which a discount factor can be applied; this greatly changes comparative GWP indexes between biofuels and other fuels [54].
— *Analytical horizon.* An analytical horizon extending into decades requires predictions about the expected cultivation period and post-cultivation LUC, decisions on how post-cultivation LUC emissions should be credited, and assessment of the time value of benefits and costs. Benefit–cost analysis brings with it the need to settle on a reasonable damage function and an appropriate discount rate as well. The UN Framework Convention on Climate Change has decided to apply a 100 year time horizon for its political decision making (e.g. Kyoto Protocol), whereas the US EPA considered both 30 and 100 year time horizons, finally using 30 years for RFS2. Policy-makers may find it appropriate to focus on more certain, near-term climate impacts, in which case a short horizon for fuel warming potential (FWP)[8] is sufficient. For short analytical horizons, discounting has little effect and post-cultivation LUC occurs beyond the system boundary [54].

[7]The 30 year time horizon was also used by Searchinger *et al.* [1] and PAS 2050.
[8]FWP is defined as the ratio of the cumulative radiative forcing caused by the life cycle GHG emissions from a biofuel relative to that of its fossil substitute. Where discounting is desired, O'Hare *et al.* [54] propose an 'economic' version of the FWP, defined as the ratio of the net present values of the cumulative radiative forcing from the two fuels. Any positive discount rate magnifies the importance of early emissions.

2.4.1. Can indirect land-use change uncertainty be truly captured?
In summary, current model estimates of CI for biofuels may be biased downwards or upwards if not accurately including all indirect emissions (without much more research, this issue will not be easily solved). These include:

— the inability of economic models to recognize unmanaged, therefore un-priced and untraded, land; much of this land is high-carbon-stock forest, and forest has been a important source of cropland [55];
— over-estimates of price-yield elasticities for crops [56];
— the assumed production period over which ILUC is 'amortized' may be too long or too short for some fuels;
— decreases in livestock GHG emissions may offset a large fraction of ILUC emissions.

To date, no models have presented a systematic description of the uncertainty (ideally in the form of confidence intervals or a probability density function (PDF)) implied by the variation or uncertainty in their parameters. A meta-analysis of model and parameter uncertainty in ILUC estimates is presented in Plevin [57]; the important implications of this analysis are the wide error bands associated with any estimate of ILUC for a given fuel, and the asymmetry of the PDF implied. In turn, this asymmetry suggests that the ILUC assigned to a fuel for administrative purposes by policy is probably not the modal value of the pdf but something higher, depending on the cost of error in this assignment (the difference between the real but unknown ILUC of a fuel and the value inferred from a model or models and used in policy implementation). A systematic treatment of uncertainty in this context is a matter of ongoing research; O'Hare [58] sketches such an analysis as a problem in decision theory. Instead of stepwise approaches of uncertainty, a 'corridor' approach could be employed to estimate cumulated ranges of key assumptions in scenarios, and in this way results can be derived without referring to individual 'events' of uncertainty [9].

3. DISCUSSION

ALCA and CLCA are used to answer different questions and therefore provide variable results which must be interpreted carefully. ALCA models direct and upstream (vehicle tank-to-field) energy consumption and direct and upstream emissions throughout a fuel supply pathway; this process poses unique constraints within an analysis of a biofuel production system that is inherently complex. ALCA allocates energy and emissions between the fuel and any co-products, and the results aim to reflect the average total emissions associated with a unit of production. Allocation choices are as critical as system boundary choices, as the value ascertained from energy (mass or carbon) content versus substitution is divergent. Through evaluation of scenarios, over time, and data that can adequately capture a dynamic system, direct emissions can be estimated over geographical areas. This is contrary to the CLCA analysis where land

intersects with an exchange of land elsewhere, and complex global commodities markets.

Therefore, CLCA is much larger in scope than ALCA and naturally is accompanied by uncertainty, owing to the complexity of the real world systems, their interconnections and the scope of the CLCA. This scope includes the total emissions from fuel production, plus all indirect effects that cascade over time, resulting from economic effects. CLCA includes emissions that are within the fuel pathway's system boundary plus those that are outside that boundary, e.g. 'anywhere in the world'. Without adequate time series, and scenarios that are sensible, models can only accomplish an isolated evaluation in ALCA, whereas the results of a CLCA depend on a combination of models and data sources used to calculate an overall CI value for an uncertain set of variables representing a complex orchestration of economic behaviour.

ILUC from biofuels has caused an intense global debate which developed over a relatively short timeframe (under 4 years). It has focused on policies that have been developed as an impetus to change and on what effects need to be measured in lieu of rapidly changing biofuel policies. Using LCA to model direct effects within the production chain at a given place in time is inherently difficult. As the scope is expanded to include policy choices, additional data on global markets, rates of penetration and effectiveness of new technologies on those markets must reflect the time scale of emissions (e.g. time horizon) resulting from an overall perturbation of global commodity markets. Indirect analyses incorporate critical choices and can include such effects through partial and general equilibrium modelling.

3.1. Recommendations

As they develop, biofuel mandates should benefit in their presentation of expanded scenarios, rather than just intermediate results, which vary among fuel LCA models making the comparison, disaggregation and uses of these values very difficult. Measuring the indirect effect of one production cycle in one country and assuming a 1 : 1 displacement effect on another land mass elsewhere is not tenable. For example, cattle stocking rates are much higher in South American production systems than for US soy production [59]. However, multiple models and combined analysis results are incorporated into policies such as the LCFS and RFS2 in the USA, which aim to evaluate the heterogeneity of indirect effects. This limitation can only be analysed more carefully through several sequential model runs, inclusive of a range of scenarios (e.g. elasticities on crop yields) and perhaps even a range of results rather than one final number (e.g. the 'risk adder' used in the LCFS framework). The European Union (EU) has taken a different approach whereby indirect effects are modelled as reference scenarios, thereby focusing on technical improvements by supplying a combination of off-limit areas, e.g. high carbon stock and biodiversity areas, and exemplifying regional cases where biofuel operators can benchmark improvements.

Perhaps more fundamentally, CLCA requires anticipating time-sensitive, nonlinear parameters (e.g. the effectiveness of existing and future policies designed to control and manage deforestation and afforestation, such as carbon policies focusing on reducing emissions from deforestation and degradation), including links with livestock management policies that can be governmental, industrial or non-governmental in nature or a combination, as with Brazil's soy and livestock moratoria.

Regulators and policy-makers should clearly distinguish between the best-available estimate of fuel CI for use in purely physical substitution (MJ/MJ), GHG emission comparisons and the CI estimate that 'should' be used in any given policy implementation [13]. Among the considerations separating these different values of CI are the different time profiles of GHG emissions [54] and the asymmetry of the distribution of the CI value [5]. More generally, the uncertainty associated with all, or any, estimates of ILUC is not random, nor is any best distribution estimate centred at zero; biofuels policy should not implicitly act as though it is by 'ignoring' ILUC on grounds of uncertainty. Much resultant policy (economic, health and safety, environmental and more) is made in the face of uncertainty and accommodates it using a range of instruments.

Recommended improvements to the CLCA framework include methodological choices, and alignment of policies that differ in model approaches, in addition to parametric standardization. Carbon stock calculations are critical to evaluate, and update, and should include a more careful evaluation of the accounting of above- and below-ground biomass, inclusive of root measurement. Various metrics as applied to fuel LCA lead to the presentation of widely varying WTW and well-to-tank results.[9] The inputs to fuel LCA models are often difficult to relate to operational data and parameters with physical meaning. Fuel LCA models tend to deal with energy inputs and efficiency while real world plant operators may deal with scf, barrels, kW, $ and many other units of commerce. The result is that the input values to models (both WTW and LUC) are often distant cousins of the physical parameter being measured.

In the absence of certainty on the magnitude of ILUC, more research and policy measures should be focused on mitigating the risk of ILUC. Potential measures include investing in agricultural research and development to increase yields of energy and food crops, protecting high-carbon-stock land, identifying and cultivating degraded land (or, conversely, more adequately matching crops to land productivity), and disseminating best agronomic practices. Policy instruments should reflect the nature of this uncertainty and target practical ways to encourage the private sector to minimize GHG emissions and wider environmental impacts throughout the life cycle (ALCA) and penalize damaging behaviour (through ALCA and CLCA). ILUC poses a serious challenge in this respect, primarily owing to the inability of suppliers of either biofuels themselves or the feedstocks they are likely

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[9]Lifecycle or 'well-to-wheels' (WTW) emissions refer to the 'well-to-wheel' and 'tank-to-wheel' (TTW) emissions, therefore incorporating 'well', e.g. farming through to the fuel combustion process.

to be made from to fully address indirect impacts. However, CLCA provides an opportunity to understand and manage macro-systems-level impacts, both positive and negative, and modify policy as a result.

4. CONCLUSION

Substantial differences currently exist between the nature of the instruments being deployed between the world's major markets for biofuels, with divergent priorities emerging between climate-change mitigation and adaptation, energy security, food security and rural development. New tools are under development to understand, measure and monitor land use and land-use change and the under- and over-lying carbon stocks. This paper has explored the potential use of a hybridized approach to estimate CLCA and ALCA impacts from biofuels and has highlighted differences in approach between the US and EU policy-makers. In so doing, it has classified the roles and opportunities for using ALCA versus CLCA. This is an important novel approach to hybridizing the two, in order to provide a better understanding of the systemic consequences on the global provisioning system of policies designed to stimulate one-specific market or production system. Work is urgently required to understand and standardize the accounting frameworks before serious, and damaging, distortions are introduced to the trade in biomass for biofuels, for heat and electricity, and by extension to the food-based commodity and emerging biomaterials and biochemical markets.

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