WHITE PAPER: GLOBAL BEST PRACTICES IN LOW-CARBON FUEL STANDARDS

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White Paper: Global Best Practices in Low-Carbon Fuel Standards

This white paper has been prepared by Pollution Probe and Savant Technical Consulting as part of a series of publications the authors prepared to provide a critical review on issues related to greenhouse gas accounting, indirect land use change, other indirect effects, and sustainability criteria for low-carbon fuel standards. A technical brief, released in September 2017 is available at the enclosed link.[1]

ABOUT SAVANT TECHNICAL CONSULTING

Savant Technical Consulting is a small, research-driven consulting company, built by world-leading experts in the fields of microbiology and biotechnology, process and bio-process design, and life cycle assessment (LCA)

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ABOUT POLLUTION PROBE

Established in 1969, Pollution Probe is a national, non-profit organization that exists to improve the health and wellbeing of Canadians by advancing policy that achieves positive, tangible environmental change. Pollution Probe has a proven track record of working in partnership with industry and government to develop practical solutions to environmental challenges.

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[1] http://www.pollutionprobe.org/publications/global-best-practices-in-low-carbon-fuel-standards/

Executive Summary

A low-carbon fuel standard (LCFS) is a regulatory tool that requires mandatory reductions in the annual average life cycle greenhouse gas (GHG) intensity of fuels sold within a given jurisdiction. LCFS-type policies focused on transportation fuels have been in place in California, British Columbia, Oregon and Europe since the late 2000s, and are joined by recent efforts including a proposed Canada-wide Clean Fuel Standard and Ontario's Renewable Fuel Standard for Gasoline. The Clean Fuel Standard, unlike the other policies, would cover not just the transportation sector but the building and industry sectors as well. The current white paper provides guidance to developers of LCFS policies by providing a critical review of key issues related to GHG accounting, life cycle framework considerations and approaches to quantifying emissions from land use change and other indirect effects as well as sustainability criteria.

Unlike the physical properties of a fuel (e.g., energy density), a fuel's life cycle GHG emissions cannot be directly measured from the final product, and instead must incorporate the context of how the fuel is produced. There is no single correct method to account for the life cycle GHG intensity (generally referred to as carbon intensity in the policies) of a fuel. Policy developers must carefully consider the consequences of the calculation methodology on the relative carbon intensity of fuels, and accordingly specify consistent procedures, such as what processes are included within carbon intensity calculations (e.g., how far up the fuel's supply chain), or how to apportion emissions among products in a multi-product system, and so on. These methods should include adjustments for differences in efficiency among fuels used for any given end-use, and should address contentious issues such as the extent to which biomass combustion is considered carbon neutral. Even with consistent policy framework procedures in place, there are likely to remain important uncertainties in life cycle carbon intensity estimates that regulators can moderate by specifying data quality and reporting/validation requirements, setting minimum GHG reduction thresholds for fuels to qualify, and encouraging fuel production pathways using feedstocks and production processes with the highest probability of achieving overall life cycle emissions intensity reductions.

Existing LCFS policies are primarily based on traditional attributional life cycle emissions accounting. A potential shortcoming of the attributional approach is that it typically assigns average status quo emissions to existing or potential products, and so is not necessarily predictive of how emissions will change if production volumes (e.g., of fuels or electricity) are increased or decreased. Consequential life cycle approaches, in contrast, aim to take into account the market-mediated effects of increased/decreased production or consumption that may arise from an LCFS policy, including indirect land use change and other indirect effects (e.g., quantities and mix of fuels consumed). In principle, these indirect effects can be included directly within the LCFS framework, but quantifying them is challenging, and existing studies differ as to their magnitudes. Other options available to LCFS regulators may include setting GHG reduction thresholds before fuels can qualify, structuring additional incentives (e.g., minimum/maximum LCFS credit for certain fuel categories), or adopting complementary policies aimed at mitigating certain undesirable indirect effects. Regardless of the approach taken, it is generally prudent for LCFS regulators to monitor the latest evidence on indirect effects and adjust the policy as appropriate. Such adjustments require care to avoid creating undue regulatory uncertainty that could stifle investment in low-carbon fuels.

Finally, although LCFS policies are focused on reducing GHG emissions, past policy experience has shown that a focus on a single impact can result in unintended negative consequences. Non-GHG sustainability criteria are challenging to integrate within an LCFS framework, but are important to consider. Through judicious fuel incentives or restrictions, together with complementary policies outside the LCFS, regulators can enhance the possibility of positive outcomes and increase the likelihood that appropriate fuels are incentivized.

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Abbreviations and Acronyms

CFS – Clean fuel standard

CO_2e – Carbon dioxide equivalent (in climate forcing intensity)

COP – Coefficient of performance

DDGS – Distillers' dried grains with solubles

EC – European Commission

EEM – Economic equilibrium model

EER – Energy economy ratio (or energy effectiveness ratio)

EPA – Environmental Protection Agency (US)

FAPRI – Food and Agricultural Policy Research Institute

FASOM - Forest and Agricultural Sector Optimization Model

FQD – Fuel Quality Directive (European Union)

GHG – Greenhouse gas

GLOBIOM-EU - Global Biosphere Management Model (European Union)

GREET – The Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation Model

GTAP-BIO – Global Trade and Analysis Project with Biofuels

ILUC – Induced land use change. Acronyms for land use change are used inconsistently in the literature. 'ILUC' has been used to represent the international component of indirect land use change, all indirect land use change, or all induced land use change (including direct and indirect).

ISO – International Organization for Standardization

kWh – Kilowatt-hour

LCA – Life cycle assessment

LCFS – Low-carbon fuel standard

MIRAGE-BioF - Modeling International Relationships in Applied General Equilibrium - BioFuel

MJ – Megajoule (1,000,000 Joules)

RED – Renewable Energy Directive (European Union)

RFS – Renewable fuel standard (US)

1 | INTRODUCTION

A low-carbon fuel standard (LCFS) is a regulatory tool that requires mandatory reductions in the annual average greenhouse gas (GHG) intensity of fuels sold within a given jurisdiction. Typically, fuel providers earn credits or generate deficits based on the GHG intensity (hereafter, carbon intensity[2]) of fuels they sell within a given jurisdiction; these credits can be traded on an open market. The Canadian government is in the process of designing an LCFS-type policy, the Clean Fuel Standard (CFS) (Government of Canada 2017). At the same time, the Government of Ontario is developing a Renewable Fuel Standard for Gasoline, which aims to reduce GHG emissions from gasoline in the province by 5% by 2020. The Ontario Standard is intended to complement the federal CFS and compound total GHG reductions from the transportation sector. The present white paper offers guidance on the design of such policies, particularly with respect to key environmental aspects.

LCFS and related policies have been implemented in a number of jurisdictions in North America and Europe, and have been the subject of analyses relating to their effectiveness and design (e.g., Andress, Nguyen, and Das 2010; Scott 2017; Yeh et al. 2016; Yeh and Sperling 2010). Important LCFS and related policies are summarized in Table 1. These existing policies are conceptually similar, sharing a focus on reducing the "life cycle" carbon **intensity** (per unit of fuel energy) of transportation fuels, including emissions associated with their production, distribution, and use, rather than reducing **absolute** GHG emissions from the regulated sector(s). To ensure absolute reductions in GHG emissions, such policies must generally be coupled with additional efforts to reduce total fuel consumption. Details of those efforts are beyond the scope of this report. Among the policies summarized in Table 1, there are substantive differences, in particular with respect to: 1) the framework for calculating the life cycle carbon intensities of fuels; 2) consideration of indirect policy consequences; and 3) inclusion of non-GHG environmental impacts.

In this white paper, we undertake a critical review of key issues related to GHG accounting and approaches to quantifying emissions from land use change and other indirect effects, as well as sustainability criteria for LCFS policies. Although a primary goal of this white paper is to inform Canadian CFS development (see Box 1), most of the discussion applies broadly to any LCFS-type policy.

[2] In this report, we use the term "carbon intensity" for consistency with language used in existing and proposed LCFS policies. Nevertheless, we prefer the more general term "GHG intensity", as it better reflects the non-carbon gases (e.g., N_2O) included in GHG emission accounting.

TABLE 1. OVERVIEW OF EXISTING LCFS AND CLOSELY RELATED POLICIES

Jurisdiction & Policy Name[a]	Goal	Fuel Eligibility	Land Use Change Included?	Other Sustainability Criteria[f]
British Columbia LCFS	10% reduction in carbon intensity of transport fuels by 2020; 15% by 2030 (relative to 2010)	All fuels used in transportation are eligible	Direct[c] only	No
California LCFS	10% reduction in carbon intensity of transport fuels by 2020 (relative to 2010)	All fuels used in on-road motor vehicles are eligible	Direct and Indirect[d] Based on GTAP-BIO model[e]	No, but includes an unspecified requirement for future sustainability provisions
Oregon Clean Fuels Program	10% reduction in carbon intensity of transport fuels by 2025 (relative to 2015)	All fuels used in on-road motor vehicles are eligible		No
EU Fuel Quality Directive (EU FQD)	6% reduction in carbon intensity of transport fuels by 2020 (relative to 2010)	Fuels used for road transport and non-road mobile machinery. The fuel must meet a GHG reduction threshold of 35% relative to the fuel it replaces; increased to 50% (2017) and to 60% (2018)	Direct only. Indirect emissions must be reported, but are not included in calculations. MIRAGE-BioF and GLOBIOM-EU models have been used.[e]	Yes, restricts biomass feedstock from land with high biodiversity or high carbon stock; limits biofuels produced from cereal, sugar, and oil crops
US Renewable Fuel Standard (RFS2)[b]	36 billion gallons of biofuels produced annually by 2022	Biofuels used in motor vehicles or in non-road engines. The biofuel must meet a GHG reduction threshold of 20% to 60% depending on fuel types (relative to fuel it replaces – gasoline or diesel)	Direct and Indirect Based on FASOM and FAPRI models[e]	Yes, excludes biofuels from non-agricultural land or from trees on federal land. Rulemaking also analyzed, but did not account for, economic impacts, and likely impact on non-GHG pollutants and water use.

[a] Citations (year refers to date of initial statute): BC 2008; CARB 2006; EC 2009; Oregon 2009; US EPA 2007

[b] This policy is not an LCFS policy but includes relevant GHG accounting methods and land use change modeling.

[c] Direct land use change refers to land converted specifically to grow the biofuel in question.

[d] Indirect land use change refers to land conversions that occur throughout the global agricultural system in response to biofuel production.

[e] Land use change models are described further in Section 3 and Appendix I.

[f] Refers to whether the policy includes sustainability criteria other than GHG emissions.

BOX 1 PROPOSED CLEAN FUEL STANDARD FRAMEWORK

In December 2017, Environment and Climate Change Canada released a proposed Clean Fuel Standard (CFS) framework for consultation (Government of Canada 2017). The proposed CFS framework is conceptually similar to existing low-carbon fuel standard (LCFS) policies, wherein fuels will be required to achieve a reduction in carbon intensity relative to baseline fuels. The proposed CFS is unprecedented in that it will regulate the carbon intensity of fuels used in transportation, buildings and industry, rather than in the transportation sector alone. Although this choice will undoubtedly introduce additional administrative complexity, appropriate credit-trading between regulated sectors and fuel streams can offer regulated parties a greater range of compliance options, resulting in lower compliance costs. According to the CFS framework, separate carbon intensity targets will be set for solid, liquid and gaseous fuel streams, with potential for the further grouping of fuel types within streams (e.g., liquid transportation fuels). At the time of release, it is not clear how the CFS will account for different forms of fuel that could be used to deliver the same energy service; for example, transportation applications could use liquid fuels (gasoline, diesel, various biofuels and alternative liquid fuels), gaseous fuels (natural gas, biomethane, hydrogen) or electricity.

The proposed CFS framework implicitly defines an "attributional" life cycle assessment approach, one that includes only direct emissions and excludes indirect emissions[3] such as those arising from indirect land use change (see Section 3 for further discussion on these topics). Some other jurisdictions have similarly excluded indirect effects, including the EU Fuel Quality Directive (the policy requires indirect emissions to be reported but they are not included in carbon intensity calculations) and the British Columbia LCFS, whereas US LCFS policies (California, Oregon) and the US RFS2 have explicitly included estimates of indirect land use change impacts.

[3] For this report, direct emissions for fuels are considered to be use-phase and supply chain emissions associated with production, combustion and distribution of the fuel. Indirect emissions encompass the market-mediated effects of increased/decreased production or consumption arising from fuel production (e.g., increased demand for agricultural feedstock used in biofuel production; change in demand for conventional fuels due to price shifts).

2 | LIFE CYCLE FRAMEWORK

The goal of an LCFS policy is to reduce the GHG emissions per unit of fuel energy consumed (e.g., grams of carbon dioxide equivalent per megajoule of fuel (g CO_2e/MJ fuel)), while accounting for emissions associated with the production, distribution and combustion ("life cycle") of the fuel. Unlike the physical properties of a fuel (e.g., energy density, sulphur content), a fuel's life cycle GHG emissions cannot be directly measured from the final product, and instead requires knowledge of where the fuel came from, how it was produced, inputs consumed and their production impacts, and so on. For this reason, there is no objective test of life cycle carbon intensity, but rather a series of guidelines and partially standardized protocols for calculating and modeling the GHG emissions associated with a given fuel. Further, indirect effects of changes in fuel production/use that arise due to LCFS policies can influence net GHG impacts (e.g., changes in total quantities of fuels consumed; changes in global land use; or changes in the electricity sector to accommodate vehicle electrification). The inability to measure direct life cycle emissions, together with the difficulty of characterizing indirect emissions, are sources of controversy that have led some authors to question the GHG mitigation potential of LCFS-type policies (e.g., Plevin, Delucchi, & O'Hare, 2017).

Given the aspiration of LCFS policies to be 'performance-based,' they must draw on established tools for conducting life cycle assessment (LCA). The International Organization for Standardization (ISO) has adopted a series of guidelines outlining basic requirements for conducting an LCA (ISO, 2006a, 2006b). Though useful, these guidelines provide only a set of minimum requirements, with many aspects of the analysis remaining subject to the analyst's judgment. In order to achieve consistent assumptions between different fuel types and analysts, standard procedures are defined within LCFS policies, including, but not limited to those covering the following dimensions:

- **System boundary** - defines which processes are included within the carbon intensity calculations (e.g., how far up the supply chain? Are auxiliary processes, such as fuel storage, included?). We note that any LCA must invariably set boundaries on the study (e.g., are emissions from the construction of the fuel production plant included in the boundary? What about the equipment used to build that plant? The raw materials used to build that equipment? The fuel required for plant operators to commute to work?). Typically, scoping estimates or past studies and LCA conventions may reveal which processes are expected not to contribute appreciably to the final carbon intensity value. Where ambiguity exists, it is advisable to take a conservative approach and include additional processes, thereby capturing more of the system impact and reducing the risk that omitted emissions will overturn the benefits of an LCFS.

- **Co-product treatment procedures** - dictate how emissions are to be apportioned among products in a multi-product system (e.g., are the emissions from petroleum refining to be divided among products based on mass? Energy content? Economic value?)

- **Impact assessment method** - specifies which gases are included in the carbon intensity values, and how they are converted to common units of CO_2e

- **Baseline fuel pathways** - specify the pathway(s) against which alternative fuel pathways are evaluated

- **Functional units** - specify the basis on which fuels are compared (e.g., per MJ of heat generated? Per vehicle kilometre travelled? Per kWh of electricity produced?)
- **Data quality requirements** - provide guidance on factors such as how recent input data must be, when process specific (vs. industry average) data is required.
- **Other methodological decisions** - for example, is biogenic carbon dioxide treated as a net emission source or as carbon neutral (e.g., Downie et al. 2014; Wiloso et al. 2016), and to what extent are indirect emissions taken into account for different fuel sources?

The following sections illustrate some of the complexities and provide recommendations related to a subset of the above dimensions. Indirect effects are considered in Section 3.

2.1. BASELINE FUEL PATHWAYS, CARBON INTENSITIES AND FUNCTIONAL UNITS

Baseline fuel pathways, including all feedstock and fuel production activities and associated carbon intensities, must be selected to represent "business as usual" baseline fuels against which the alternative fuels and production processes' carbon intensities will be compared. It is important that baseline fuel carbon intensity values are accurate, that the data/methodologies used in their development are transparent, and that they are defined to provide incentives for all fuel producers/regulated bodies to reduce GHG emissions. Ideally, baselines would consider the 'marginal' carbon intensity values, which reflect those fuels most likely to be displaced by lower carbon fuels, rather than a simple system average (Siler-Evans, Azevedo, and Morgan 2012; Wallington, Anderson, De Kleine, et al. 2016), although achieving this in practice is challenging.

Carbon intensity targets and fuel evaluation methodologies must be designed to ensure they consider the function that can be delivered by a fuel pathway to avoid inadvertently favouring or disadvantaging a particular pathway. For example, current and future vehicle technologies will differ in terms of their efficiencies (fuel consumption) when using certain fuels, and so a direct comparison of fuels on an energy basis (i.e., per MJ of fuel) would not accurately reflect their relative life cycle GHG emissions. As a specific example, diesel vehicles are generally more efficient than gasoline vehicles, achieving 10-20% lower fuel consumption than gasoline vehicles per unit of energy (U.S. Department of Energy 2018), resulting in lower overall GHG emissions. However, diesel has a higher carbon intensity than gasoline by approximately 7% on an energy basis (BC Ministry of Energy and Mines 2017). Adopting this carbon intensity without any efficiency adjustment could incentivize a switch from diesel to gasoline (based on carbon intensity), even though switching in the opposite direction (from gasoline to diesel) would actually have a better impact on overall GHG emissions. Similarly, ethanol may enable more efficient vehicle engines through higher octane ratings (Anderson et al. 2012), and so comparing fuels only based on g CO_2e/MJ may underestimate the benefits of bioethanol. As a result, existing LCFS policies typically make adjustments to carbon intensity values when calculating the quantity of credits generated by different fuels. A sample procedure is illustrated in *Box 2*.

BOX 2 ILLUSTRATING EFFICIENCY ADJUSTMENTS IN LCFS CREDIT GENERATION – ELECTRICITY

Electric vehicles and heat pumps can provide transportation and heating services, respectively, at a generally higher efficiency than other fuel pathways. In existing LCFS transportation fuel regulations, a correction factor, termed the ***energy economy ratio*** (California LCFS) and ***energy effectiveness ratio*** (BC LCFS) is used to account for the relatively lower fuel consumption (higher fuel economy) of electric vehicles to determine LCFS credits using the following calculation:

$$Credits = \left(CI_{standard} - \frac{CI_{electricity}}{EER}\right) \times Energy\ Density\ \times EER\ \times Electricity\ units\ \times 10^{-6}$$

where $CI_{standard}$ is the baseline fuel (gasoline, diesel) carbon intensity (g CO_2e/MJ); $CI_{electricity}$ is the electricity carbon intensity (g CO_2e/MJ); the ***Energy Density*** of electricity is 3.6 MJ/kWh; ***EER*** is the energy economy/effectiveness ratio; and ***Electricity units*** is the consumption of electricity for transport applications (kWh). EER is defined as the service provided by an alternative fuel (e.g., km of vehicle travel per MJ electricity) divided by the service provided by the traditional fuel (e.g., km of vehicle travel per MJ gasoline). An equivalent formula (with appropriately adjusted EER) can be used for calculating the credits or deficits generated by any fuel source for which the efficiency of use differs from the baseline fuel it replaces.

In a broader policy that includes non-transportation fuels, like the CFS, a similar approach should be applied to space heating and other fuel use. For example, electric heat pump systems provide a greater quantity of heat per unit of energy input than many competing space heating technologies. The choice of EER must vary not only by fuel type, but also by end-use (e.g., electricity used in vehicles vs. electricity used in heat pumps) and also sometimes by surrounding conditions. In the case of heat pumps, the coefficient of performance (COP), which measures the ratio of heat output to electricity input, is dependent on operating conditions: COP decreases as the difference between hot (indoor) and cold (outdoor) temperatures increases. As such, a range of energy economy/effectiveness factors would have to be used to reflect regional climate differences.

2.2. DATA REQUIREMENTS AND LCA MODEL UNCERTAINTY

By including impacts across the full life cycle, LCA aims to comprehensively identify and quantify sources of environmental impact. However, variability and uncertainty in data, as well as differences in modelling approaches, can result in LCA models returning different results. Selection of data to best represent the system in question is critical, taking into account potential sources of variability (data may vary with time, location, or producer) and uncertainty (data may be aggregated, incomplete, out of date, not representative, or not available). Given the large number of required inputs and their associated variability and uncertainty, LCA results will always have some degree of uncertainty.

There exist a number of publicly available LCA tools, including well known models such as: GHGenius, developed by Natural Resources Canada (NRCan) and (S&T)[2] Consultants (Canada) ((S&T)[2] 2018)); The Greenhouse Gases, Regulated Emissions, and Energy use in Transportation Model (GREET), developed by Argonne National Laboratory (US) (Argonne National Laboratory 2017); and BioGrace (EU) (IFEU 2015), among others. Although the three listed models focus on transportation fuels, their structures are reasonably suited for modeling fuels in other sectors as well. An appropriate tool (if any) should only be selected after conducting a critical examination of the data, structure and assumptions embedded in such models. The model should also be up to date and geographically relevant, with clear plans specified in the policy to ensure that it remains current. It is also incumbent on policy makers to ensure that any model or study used within an LCFS is highly transparent, publicly available, and well documented, including a detailed user manual (facilitating review, validation and verification of data and modeling assumptions).

Models as compliance tools typically allow fuel producers to enter primary data for key inputs to ensure results are representative of current processes. Where primary data are unavailable, model defaults can be used, although these default values require regular updates for accuracy. Default values may or may not be appropriate for use in LCFS. As an example of default values, GHGenius widely employs time series to estimate default values (agricultural feedstock yield; biofuel production inputs and yield), with values from a reference year extrapolated to the present and future based on observed historical trends. Although some base data in the most recent publicly available version (v5.0a) are about 20 years old (as of 2018), the fitted trends typically account for more recent data, allowing the model to calculate values representative of more recent years. Data in GHGenius v5.0a for production inputs to corn ethanol use 1999 as a base year, with fitted reductions in electricity and natural gas requirements (2% and 1.8%/yr, respectively) and improvement in ethanol yield (0.2%/yr), which leads the model to calculate correspondingly lower GHG intensity with each passing year. As such, GHGenius' default assumptions for current corn ethanol production are considerably different from the base data (~30% reduction in process energy consumption), which partly reflects actual historical reductions in process energy use since 1999, and partly reflects the assumption that these trends continue to the present. Modeled trends allow users the ability to forecast future performance, but still require procedures for on-going updating of carbon intensity values with regularly collected data to ensure modeled results remain current.

Existing LCFS policies use default values for fuel carbon intensities as a means to partially address data variability/uncertainty and to encourage fuel suppliers to utilise actual data to calculate compliance with carbon intensity targets. Default values related to feedstock cultivation, fuel processing, and transport/distribution provide a common basis for all producers under an LCFS policy. By setting default values artificially high (e.g,. BC Ministry of Energy and Mines 2017), fuel suppliers have an incentive to demonstrate lower carbon intensity of their fuels by reporting actual data to determine fuel carbon intensity. The EU Fuel Quality Directive, for example, utilises default GHG emission values that are approximately 20-25% greater than typical emissions expected for particular fuel pathways (EC 2015). Existing policies, including the EU policy, also use minimum GHG reduction thresholds, which can increase the likelihood that supported fuels achieve overall reductions in carbon intensity despite uncertainty and variability in GHG emissions. At the same time, thresholds can exclude cost-effective fuels that can reduce GHG emissions, albeit by a smaller amount.

2.3. LIFE CYCLE FRAMEWORK: RECOMMENDATIONS

In summary, key recommendations for LCFS policies related to life cycle framework considerations include:

- Recognizing that LCA is not an exact science, the life cycle framework and associated data require periodic review and adjustment to ensure that the LCFS is having its intended effect
- Developing a comprehensive life cycle framework that can be used for all fuels to ensure consistency between carbon intensity values
- Carefully selecting and regularly updating baseline values, with consideration given to trends in marginal fuel sources
- Developing a framework to update processes and fuel pathways on a regular basis
- Incorporating adjustments for efficiency of fuel-use, rather than regulating on a g CO_2e/MJ basis alone
- Ensuring that any LCA models employed are transparent, well documented, and use appropriate and current data
- Developing explicit strategies to account for uncertainty within carbon intensity values such as:
    - o Setting strict measurement, verification, and reporting requirements for company-level and process-specific data inputs
    - o Requiring that LCA models include explicit uncertainty analysis (e.g., including ranges or confidence intervals for carbon intensity values), to ensure the selected regulatory carbon intensity value is truly representative of the fuel in question
    - o Providing additional incentives for fuels that have the highest probability of achieving GHG emission reductions or contributing toward the deepest cuts in GHG emissions in the long-term
    - o Setting minimum GHG reduction thresholds (relative to baseline) before fuels can qualify for LCFS credits.

3 | INDIRECT EFFECTS OF LOW-CARBON FUEL STANDARDS

Current and proposed LCFS policies are largely based on attributional LCA methods. A potential shortcoming of attributional LCA is that it assigns average status quo emissions to existing or potential products, but is not necessarily predictive of how emissions will change if production volumes are increased or decreased. In response, some researchers and policy makers have suggested a need to analyze broader system changes that occur when an existing product's production level is changed or a new product is introduced ('consequential' LCA)(e.g., Plevin, Delucchi, & Creutzig, 2014). A consequential LCA approach typically considers marginal, rather than average, data – making use of economic models to account for the market-mediated effects of increased/decreased production or consumption. A consequential LCA approach is arguably more appropriate for evaluating the consequences of an LCFS policy on GHG emissions, but its application may be limited by a) the wide range of possible indirect effects, b) methodological and data gaps for predicting market outcomes, and c) difficulties attributing system-wide changes to any specific unit of fuel, potentially creating fundamental incompatibilities with the carbon intensity approach. Existing and proposed LCFS policies are divided on whether or not to include indirect emissions – specifically those arising from indirect land use change, a key source of controversy. A particular challenge is that indirect effects cannot be measured, and thus, are model-dependent and highly variable.

Practical applications of consequential LCA can be described as a mix of attributional and consequential approaches, wherein **some indirect emissions** associated with the economy-wide responses to policies or product adoption are considered. To illustrate the treatment (or lack thereof) of indirect effects within attributional, 'mixed attributional and consequential' and completely consequential LCA we provide a simplified example in Figure 1 related to corn ethanol. Biofuels can increase or create a demand for agricultural or forestry products and thereby induce changes in global land use patterns to supply this demand, which could result in the same or a net increase or decrease in the quantity of carbon stored in terrestrial carbon stocks (soils, biomass), with potential GHG consequences. Using an attributional LCA approach, only direct emissions would be included and so potential emissions consequences of market-induced changes in land use patterns would not be captured. A mixed attributional and consequential LCA approach could include certain indirect effects that are potentially of significance. This mixed approach has been undertaken by the California LCFS and the US RFS2 to include land use change in calculations of fuel carbon intensities. We discuss indirect land use change in more detail in Section 3.1. A fully consequential approach would aim to capture all indirect effects associated with the increased biofuel production; however, the practical implementation of such an approach would be challenging.

Electricity generation is another aspect where LCFS policies could induce changes that may be important to consider. Electricity is generated from a wide range of sources including fossil fuel (coal, natural gas) and renewable sources, which is then delivered to consumers via common transmission and distribution infrastructure. Identifying the specific generation sources utilised in sectors covered by the LCFS, therefore, is not straightforward. The simplest approach, which is common to attributional LCA, is to consider the average regional electricity generation mix. This approach is currently utilised by both the BC and California LCFS policies, which assume the average generation mix is consumed by electric vehicles within the province/state. This procedure could be refined further by taking into account differences in the average electricity mix based on time of use. The average generation mix approach, however, does not consider the consequences of policies encouraging new uses of electricity (electric vehicles; more electric heating) on the electricity generation sector. An alternative, consequential approach could be taken to identify and evaluate implications of new electricity applications for aggregate electricity demand, mix of electricity sources and the efficiency of operating generation, storage, transmission and distribution assets.

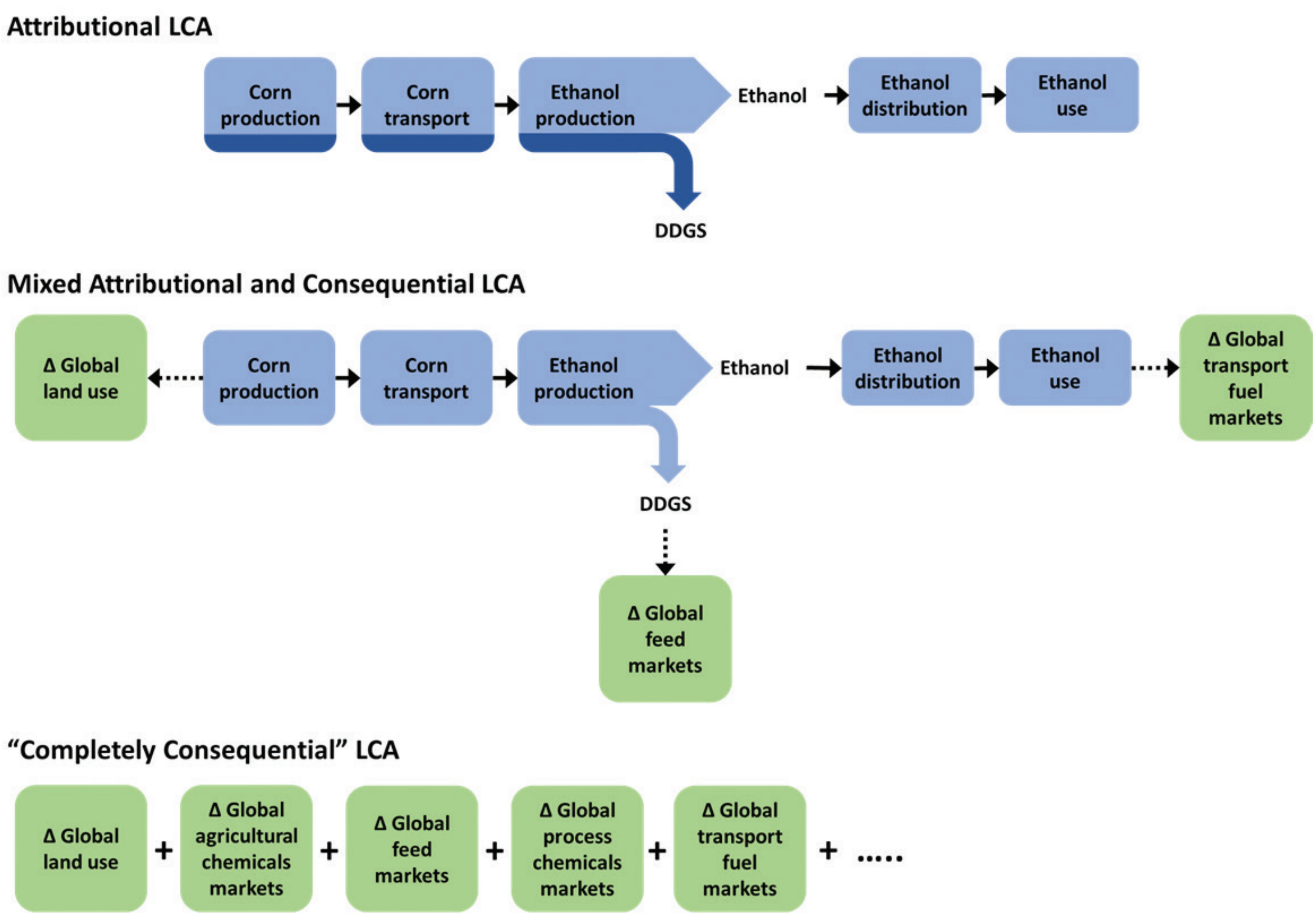

Figure 1. Scope of Attributional, Mixed, and Consequential Approaches applied to corn-derived ethanol example. Attributional LCA (top figure) is focused on the direct impacts of producing a transportation fuel, including feedstock cultivation (agriculture inputs; direct land use change effects), transportation, ethanol production, distribution and use. Co-products [i.e., distillers' dried grains with solubles (DDGS)] are allocated a share of production inputs. A Mixed Attributional and Consequential LCA approach (middle figure) expands on an Attributional LCA, which is complemented by considering a limited number of key indirect effects, including the use of DDGS in animal feed (or other) markets, induced land use change, and/or fuel market effects. All emissions – direct and indirect – are allocated to the primary ethanol product. A Completely Consequential LCA approach (bottom figure) may consider only market effects associated with demand for inputs to ethanol production (e.g., land, agricultural chemicals, process chemicals) and markets for the ethanol and DDGS products.

3.1. SAMPLE INDIRECT EFFECTS AND ASSOCIATED GHG EMISSIONS

Induced land use change is among the most prominent indirect effects, and will be discussed in greater detail below (Section 3.2) In addition, there exist a range of other potential indirect effects that may either reinforce, or offset/reverse the GHG reductions of an LCFS. With the exception of indirect land use change, indirect emissions have not yet been integrated explicitly into existing LCFS policies. Examples of indirect effects may include:

- Fuel market shuffling, whereby low carbon intensity fuels are consumed in the regulated jurisdiction, only to free up high carbon intensity fuels for consumption elsewhere (Debnath, Whistance, and Thompson 2017)
- Petroleum product mix change, whereby the petroleum refining industry responds to alternative fuels by adjusting the relative mix of products (e.g., gasoline, diesel, residual fuel oil) brought to market, with potential GHG consequences (Posen et al. 2017)
- The rebound effect, whereby lower operating cost (e.g., as is often the case with electric vehicles) encourages greater product use (e.g., more vehicle kilometres traveled) (Sorrell, Dimitropoulos, and Sommerville 2009); conversely, if the LCFS results in higher fuel prices, consumers may travel less or purchase more fuel-efficient vehicles
- The need for increased travel distance for consumers to find appropriate fuel stations for alternative fuels (Seki et al. 2018)
- The indirect fuel use effect, whereby the adoption of alternative fuels reduces the demand for fossil fuels in the regulated jurisdiction, leading to lower fossil fuel prices. For globally traded fuels like gasoline and diesel, these lower prices may result in increased international fossil fuel consumption, offsetting the GHG reductions associated with the LCFS (e.g., Drabik and Gorter 2011; Rajagopal et al. 2015; Rajagopal and Plevin 2013; Smeets et al. 2014; Thompson, Whistance, and Meyer 2011).
- Learning curve and spillover effects, whereby a government policy induces greater use of low GHG energy sources, which enables new technologies to develop and leads to lower costs (Goldemberg et al. 2004; Kobos, Erickson, and Drennen 2006). Lower costs may in turn lead to greater adoption of the low GHG energy technology, which further reduces GHG emissions (potentially even in other regions).

The above discussion provides a small cross-section of examples that illustrate the complexity of evaluating the net GHG impact of fuels (and other products) in the real world. By estimating the GHG impact associated with any of these effects and normalizing by the quantity of alternative fuel involved, an LCFS regulator can, in principle, adjust the carbon intensity of fuels to reflect indirect effects. In practice, such adjustments have only ever been applied to emissions from induced land use change (Section 3.2), and remain contentious even in that case. Regardless of whether indirect effects are explicitly included in the policy, regulators may wish periodically to study and monitor these effects to have greater confidence that the LCFS policy is resulting in the desired reduction in GHG emissions.

3.2. LAND USE CHANGE

Land use change (LUC) is defined as the change from one land use type or management approach to another (e.g., from forest to cropland; land use intensification), and could drive changes in land cover and associated changes in carbon stocks. The net GHG balance of LUC results from the change in carbon stored within biomass and soil pools, as well as other changes in land management, such as those associated with additional fertilizer use. Energy production and land use are inexorably linked. Even though direct land use for energy resources only accounts for about 2% of earth's land (Fritsche et al. 2017), increasing energy demand and shifting energy resources create new challenges in terms of land use and associated impacts.

Studies from the US (Trainor, McDonald, and Fargione 2016), Europe (Fritsche et al. 2017), and Canada (Fthenakis and Kim 2009; Yeh et al. 2010) have estimated the land use intensity for several energy systems (Appendix I, Table A1). Land use intensity refers to the area of land (m^2) required to produce a given quantity of energy, measured as land area per unit of energy. Estimated land use intensity varies greatly among systems and between studies, and is reported to range between 0.003 and 2.3 m^2/GJ for fossil fuels and between 61 and 229 m^2/GJ for agriculture-based biomass energy systems (other systems, like algae or waste-based biofuels, may have lower land use intensity). As the most land-intensive energy source, biomass has been under particular scrutiny for its potential to induce global changes in land use patterns.

In contrast to the considerable research done to evaluate the LUC-related GHG emissions due to bioenergy production (particularly biofuels), less information is available for other energy sources such as non-renewables and non-biomass renewables. For this reason, much of the discussion that follows focuses on biofuels. Some studies have historically differentiated between ***direct*** and ***indirect*** land use change, depending on whether the new emissions occur on the land that is directly used to grow biofuel feedstock or elsewhere in the global agricultural system in response to diverting land or crops to biofuel production. For the purpose of assessing the consequences of biofuel and LCFS policies, this distinction is often not useful as the net impact on emissions is the same. Increasingly, experts rely on blanket terms, such as ***induced*** land use change (ILUC[4]), to capture the emissions from all land transformations resulting from increased use of biofuels or other energy sources.

Various studies have projected ILUC emissions due to biofuel production, with estimates ranging from below 0 (i.e., removing carbon dioxide from the atmosphere) to over 200 g CO_2e/MJ (Warner et al. 2014), more than double the emissions of gasoline production and use (~90 g CO_2e/MJ), depending on the biofuel source and ILUC model employed. Values at the low end of this range can reinforce the prospective GHG benefits of biofuels, while values at the (potentially suspect) high end of this range would overturn those benefits entirely. For U.S. corn ethanol, ILUC emissions used in US policies (RFS2, California LCFS, Oregon LCFS) range from 8 to 26 g CO_2e/MJ; values are typically lower for some biofuels, like sugarcane ethanol, and higher for some biofuels, like soybean biodiesel. To our knowledge, no study has yet quantified ILUC emissions for biofuels produced in Canada, although, because the emissions value is based upon marginal effects in a global system, comparable values to those in the U.S. may result. Over time, ILUC modelling methodologies and assumptions have developed and, as a consequence, associated emissions estimates are frequently updated. For example, recent inclusion of land use intensification in ILUC models has resulted in lower assessed GHG emissions in at least one model (Taheripour, Zhao, and Tyner 2017). However, there is no broad consensus regarding the trend or magnitude of ILUC GHG emission estimates. ILUC models are challenging (if not impossible) to validate (Babcock 2015; Babcock and Iqbal 2014) and there remains a lack of consensus as to appropriate ILUC model assumptions and methodologies, which can thus lead to substantial variation in predicted ILUC values. This lack of consensus and inability to validate model results may also create a risk of legal challenges if ILUC is included as part of a regulatory system.

3.3. METHODS FOR ESTIMATING ILUC EMISSIONS

ILUC models typically estimate the amount, type, and location of global land transformations induced by fuel production. The predicted land cover changes are linked with carbon stock and emission factor databases to estimate resulting GHG emissions, as illustrated in a simplified manner in Figure 2. Other ILUC-related impacts are considered by some models, such as changes in fertilizer use and livestock production, and their associated GHG emissions. The resulting estimated emissions are then amortized over a period of time (e.g., typically 20-30 years of future production in the case of biofuels). Because ILUC stems from economy-wide interactions, the resulting emissions are attributed to different fuel types (e.g., corn ethanol), and are less dependent upon the specific production process from any given company or facility. However, if there were a set of facilities with higher product yields, they would require less feedstock, and arguably have smaller indirect effects. While use-phase and supply chain emissions from LCA models can be specific to local fuel production processes, ILUC is a characteristic mainly of the fuel type, feedstock, production area, and policy under which that fuel is produced.

[4] Acronyms for land use change are used inconsistently in the literature. 'ILUC' has been used to represent the international component of indirect land use change, all indirect land use change, or all induced land use change (including direct and indirect). This white paper uses the latter as it is the broadest definition of induced land use change.

Most ILUC modeling approaches are based on economic equilibrium models (EEMs). EEMs are used in the calculation of carbon intensity values in the US policies (Table 1) and to evaluate ILUC in Europe. EEMs start with a baseline supply and demand equilibrium across the world economy or within a set of specified interconnected markets. An economic shock is introduced (i.e., increase in demand for a biofuel due to a new policy), and the model forecasts the resulting new equilibrium. Consequently, there is a shift in global land use and management patterns to meet the new demand. EEMs estimate the quantity and geographical location of land transformations, including between land types (e.g., cropland, forest, pasture), conversion between crop types, and cropland intensification. Several factors will affect the estimated quantity and type of land use change, including assumed agricultural yield; industrial process yield of fuels from agricultural inputs; assumptions surrounding the treatment of co-products; intensification of agricultural production (i.e., induced yield increases); and trade. EEMs rely on estimated elasticities[5] to predict responses to changes in commodity prices both on the supply side (e.g., crop yields; land conversion) and the demand side (e.g., demand for a particular commodity). EEMs include both General Equilibrium Models, which cover an aggregated set of sectors to represent the entire economy, and Partial Equilibrium Models, which focus on specific sectors of interest and assume other sectors remain unchanged. There are trade-offs between the two EEM techniques: partial equilibrium models provide greater resolution on key sectors, but lack the consideration of interactions with other sectors provided by general equilibrium models. For more detail on ILUC approaches, ILUC modeling in economic models, major characteristics of prominent economic models, approaches of current policies to address ILUC, and emission factor models see Appendix I.

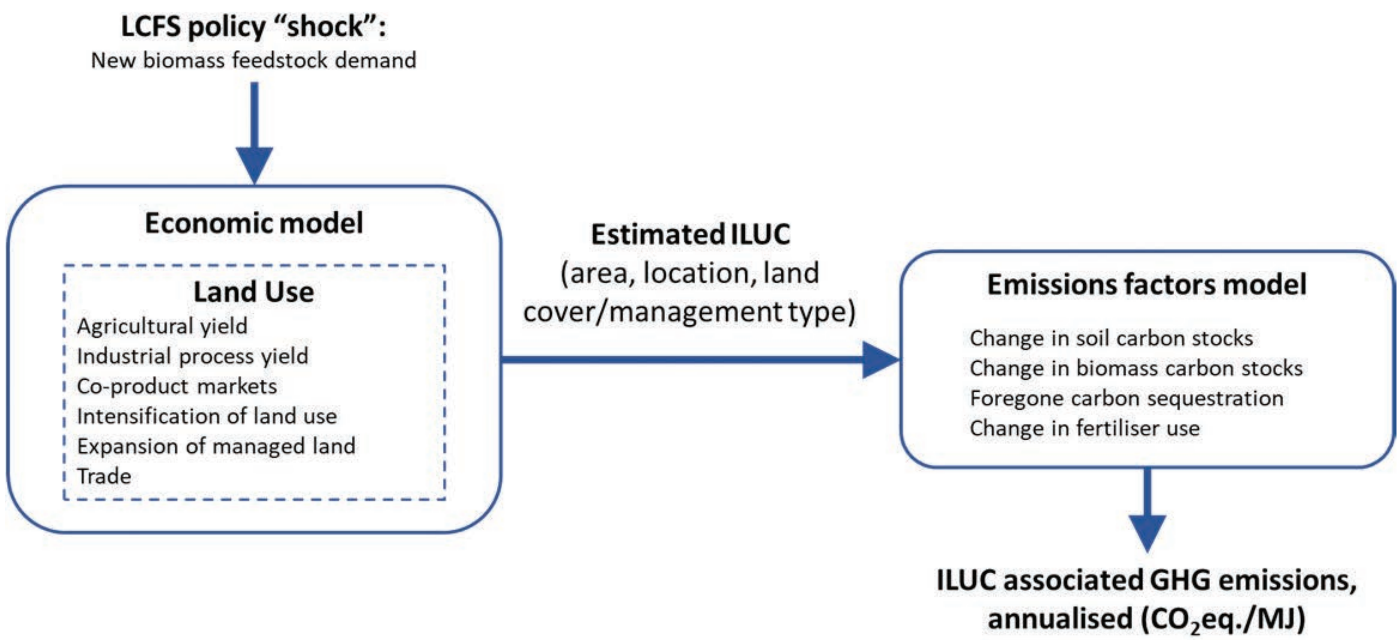

Figure 2. Schematic representation of how economic equilibrium models (EEMs) are used to estimate GHG emissions/sequestration from induced land use change (ILUC). A policy 'shock' is introduced, and combined with the baseline data and model equations (a subset of which are shown in the figure) to project land use changes. These land use changes are associated with emission factors, representing the GHG emissions induced by such transformations, and annualized to produce the final GHG estimate per energy unit of fuel.

[5] Elasticities are parameters that represent the percentage change in one variable as the result of a percentage change in another variable (usually price).

3.4. ILUC UNCERTAINTY

Quantifying ILUC is complex and models rely on many assumptions; as with other aspects of LCA modeling, some inherent uncertainty cannot be avoided. Based on literature, some uncertainties and modeling decisions that have an important influence on model estimated ILUC are related to the parameters/characteristics of the market-equilibrium model [e.g., the size of the "shock" (magnitude of assumed increase in biofuel production), price/yield elasticities, co-product treatment, yields, intensification of land, trade, the relative productivity of land converted to cropland)] and of the biophysical model (e.g., emissions factors of land conversion, peat land emission factors, the lack of high-resolution land-use data on a global scale) (Plevin et al. 2010a; Plevin et al. 2015; Prins et al. 2010; Taheripour, et al. 2017; Tyner et al. 2016; Warner et al. 2014; Wicke et al. 2011). There has been clear recognition that uncertainty and sensitivity analyses should be performed to explore ranges for ILUC results (Plevin et al. 2010b; Plevin et al. 2015; Warner et al. 2014), with scenario analysis and/or Monte Carlo simulation being used to do so. However, an underlying challenge is the impracticality of empirically validating models and results. As such, while sensitivity analyses can identify parameters with strong influence on model outputs, it may not be possible to assign rigorous statistical probabilities to particular outcomes (Babcock 2015). This uncertainty does not negate the potential reality of ILUC, but does make it difficult to specify a single value for ILUC for the purpose of an LCFS regulation.

3.5. INCLUSION OF ILUC IN LCFS POLICIES

The inclusion of ILUC estimates in LCFS policies is contentious and varies in existing policies (Table 1). ILUC emissions were considered for inclusion in the EU Fuel Quality Directive; ultimately, however, while default ILUC emissions must be reported for biofuels pathways (feedstock/fuel), they are not included in calculations of a fuel's carbon intensity nor considered when determining compliance with GHG emissions thresholds. In contrast, ILUC emissions are included when determining the carbon intensity of biofuels within the California LCFS, Oregon Clean Fuels Standard, and RFS2 policies. In an attempt to address uncertainty in ILUC estimates, the California LCFS regulator (California Air Resources Board) used a scenario approach by systematically varying the values of a subset of model parameters. The ILUC GHG emissions from all scenarios were then averaged and taken to represent the ILUC factor for specific crop-derived biofuels. For corn ethanol the resulting ILUC estimates ranged from 11.2 g CO_2/MJ to 34.3 g CO_2/MJ, with an average of 19.8 g CO_2/MJ that was adopted as the regulatory value. The California LCFS has been fraught with legal challenges (e.g., (Rocky Mountain Farmers Union V. Goldstene 2011)), motivated in large part by misgivings about the inclusion of ILUC emissions (Bevill 2011). It should be noted, however, that ILUC figured in the legal argument only as part of a broader claim that the policy aimed to regulate activity beyond its jurisdiction – including ILUC, but also all out-of-state activities that contribute to the fuel's life cycle emissions (Rocky Mountain Farmers Union V. Goldstene 2011). The fact that different models and model assumptions can lead to different ILUC estimates, especially dramatic for oilseeds, may also increase the risk of legal challenges to policies that include ILUC.

Considering the Canadian context, ILUC emissions are excluded from both the BC LCFS and the proposed national Clean Fuel Standard. To date, no EEM has been developed to specifically assess ILUC impacts of Canadian fuel policies. However, existing tools could be adapted to do so (some models are already global in nature) and may be able to provide insights to Canadian policymakers for future policy revisions. Where possible, current and proposed policies should be implemented in a manner to support the development of assessment methodologies by collecting key data from fuel producers (e.g., the types, sources, and prices of feedstocks) to aid in developing and parameterizing ILUC models. Other mechanisms can be used to mitigate against unintended negative ILUC effects, without necessarily including an ILUC value in the fuel's carbon intensity. For example, the EU Fuel Quality Directive restricts biofuel production from feedstocks grown on virgin land or land with high carbon stocks. In the UK, extra incentive is provided to waste-based fuels, which encourages producers to use feedstocks with a negligible land requirement.

Although the Canadian fuel market is relatively small on the global scale (1-4% of both global production and consumption (U.S. EIA 2017)), one should not assume that the magnitude of ILUC (on a /MJ fuel basis) is also small. Indeed, ILUC is a global phenomenon and so any change in demand for land or land management practices can in principle, induce land transformations anywhere in the world. This effect can (in theory) be transmitted through changes in global commodity prices, regardless of whether Canada is directly trading with the affected countries.

Given the inherent uncertainty and difficulty validating ILUC models, some authors have argued against including ILUC within policy frameworks (e.g. Finkbeiner 2013). In response, other authors have argued that even attributional LCA models (those with no indirect effects) have difficulties with validation, and that excluding ILUC merely masks LCA uncertainty without eliminating it (e.g. Muñoz et al. 2015). Facing disagreement within the scientific community, LCFS regulators are left with a difficult choice regarding if and how to address ILUC or other indirect effects within their policies.

3.6. LAND USE CHANGE AND OTHER INDIRECT EFFECTS: RECOMMENDATIONS

Explicitly including GHG values for ILUC and other indirect effects within LCFS policies would, in principle, result in a more complete emissions accounting. In practice, there is significant uncertainty in estimating these GHG emissions implications, especially for ILUC. Inclusion of indirect effects within LCFS policies is therefore very contentious. Where possible, LCFS policies should be used to support the development of appropriate assessment methodologies through the collection of key data from fuel producers (e.g., the type, sources, and prices of feedstocks) and national/international statistics to parameterize and, ideally, to validate models of indirect effects.

Where it is not feasible to include ILUC and other indirect effects within carbon intensity values, additional mechanisms could be used to help mitigate against unintended negative indirect effects. Such steps may include:

- Setting minimum GHG reduction thresholds (relative to baseline) before fuels can qualify for LCFS credits. In addition to helping address uncertainty (see Section 2.2), this requirement can act as a hedge against the excluded indirect emissions.
- Restricting the use of fuels from certain feedstocks, such as biomass grown on land with high carbon stocks (e.g., as in the EU Fuel Quality Directive).
- Providing additional incentives related to land use intensity of fuels; for example, for alternative fuels with very low land use intensity.
- Promoting the development of energy sources (i) grown on/placed on/extracted from marginal or degraded lands, or (ii) grown by adopting practises considered to introduce a low risk for induced land use change (e.g.,*new* implementation of cover crops or double cropping, or growth of a biofuel crop during what had normally been a fallow period). For example, the EU Fuel Quality directive provides a bonus (negative emission value) of 29 g CO_2e/MJ biofuel if the feedstock is grown on previously degraded land.
- Adopting complementary policies to increase the likelihood that the entire suite of policies achieves absolute GHG emission reductions, even when indirect effects are taken into account. For example, introduction of a carbon tax has been shown to mitigate the indirect fuel use effect (Smeets et al. 2014). Likewise, international conservation efforts may help avoid impacts from global ILUC.
- Conducting ongoing investigations to monitor the potential contribution of indirect effects, and adjusting the LCFS policy as needed to ensure appropriate fuels are being incentivized.
- Collaborating with international bodies to work towards developing harmonized global standards for ILUC accounting (e.g., UN FAO, IUCN, IPCC, UNESCO, ISO).

4 | SUSTAINABILITY CRITERIA

LCFS policies are focused on reducing GHG emissions and some frameworks like the Canadian CFS have been explicit in their exclusion of non-GHG sustainability criteria. Although climate change is one of our most pressing global challenges, past policy experience has shown that a focus on a single impact has often resulted in unintended negative consequences. How likely is it that fuels viewed as attractive under an LCFS due to their low GHG intensities actually offer net overall benefits to society? Encompassing the broader concept of sustainability (environmental, economic, and social components) is critical to ensuring a movement to low GHG fuels does not result in net negative impacts.

Within the environmental component alone, there are a great number of potential impacts to consider. In the broadest sense, nearly all environmental concerns stem from different types of resource use, the results of emissions (to air, water or land), or modifications to natural environments and habitats. Numerous attempts have been made to classify these concerns into specific impact categories (e.g. Bare, Norris, and Pennington 2003; Frischknecht, Steiner, and Jungbluth 2009; Goedkoop and Spriensma 2000; Guinee 2002; Huijbregts et al. 2017; Jolliet et al. 2003), resulting in a large number of potential environmental metrics that can be included in LCA. These include contributions to climate change (the focus of an LCFS), non-renewable resource depletion, water withdrawals and consumption, acidification, eutrophication, criteria air pollutants and local air quality, ecotoxicity and biodiversity loss. Appendix II contains a more complete list, with a brief description of each of the impact categories.

There are numerous ways in which the adoption of alternative fuels may impact (positively or negatively) these other (non-GHG) sustainability categories. Potential impacts could transcend various fuel types, from renewables to fossil fuels, and some feedstock/fuel pathways will have lower/higher sustainability impacts than others. For example, some authors have raised concerns related to water consumption and water quality associated with fuel production, although the extent and magnitude of such impacts are not well understood. Recent work commenced at the Argonne National Laboratory and the US Department of Energy's Biomass Energy Technologies Office (BETO) aims to develop quantitative metrics based on water consumption analysis and hydrologic models, and evaluate sustainable water use scenarios in the production of bioenergy and bioproducts (Wu, Ha, and Xu 2017) Natural Resources Canada has recently developed I-BIOREF (Natural Resources Canada 2015), which enables comparison of biorefinery configurations and includes water withdrawal as well as other sustainability metrics.

From a process perspective, water consumption attributable to biofuels production is relatively low, depending upon the process – and reported to be roughly on par with water use attributable to petroleum fuel production according to an international study (Spang et al. 2014). More substantive impacts have been claimed if the feedstock requires irrigation. For example, in 2011, the developers of GREET calculated average water consumption for corn ethanol to range from 11-160 gallons of water per gallon of ethanol, corresponding to U.S. corn irrigation rates ranging from 3-35% of harvested crop, compared with an average of 2.6-6.6 gallons of water per gallon of petroleum gasoline, which encompasses their estimates for both conventional and oil sands production (Wu and Chiu 2011). This example is specific to corn ethanol in the United States, and does not reflect on all biofuels/feedstocks. In contrast, irrigation is currently relatively uncommon in Canada. Some crops are irrigated in Southern Alberta, Southern Saskatchewan, and Southern Ontario, yet in 2017, in Alberta only 4% of all wheat, and only 2.6% of all canola, was irrigated (Alberta Government 2017). In Ontario, only 2% of all field crops were irrigated in 2016 (Statistics Canada 2018b)(Statistics Canada 2018a). While the current impact of bioenergy on water use in Canada may be relatively small, it is important to be aware of and manage potential impacts should circumstances change. The BETO work cited above, for example, has identified various practises that can be adopted to manage water consumption and discharge, particularly from feedstock production.

Similarly, it is important to be aware of and manage other sustainability impacts of fuels, from impacts on groundwater to water quality issues like eutrophication (Alshawaf, Douglas, and Ricciardi 2016; Miller, Landis, and Theis 2007). Awareness of these potential issues becomes key to their management, such as through improved methods for wastewater treatment, those to deliver nitrogen to crops or the use of precision agriculture. Fuels (conventional and alternative) that lead to changes in land use and habitat destruction may also impact biodiversity. It is therefore important to monitor and manage such potential impacts.

Issues like fuel toxicity and impacts of fuels on air quality are also critical sustainability aspects. For example, fuels like biodiesel may exhibit lower toxicity than petroleum diesel (e.g. Khan, Warith, and Luk 2007). As another example, existing literature suggests that increased use of biofuels like ethanol and biodiesel has potential benefits for air quality (NOx, non-methane hydrocarbons, particulate matter and mobile source air toxics) and should not result in increased emissions when used in modern gasoline and diesel vehicles and when biofuel content is within the operational ranges for which the vehicles are designed (Wallington, Anderson, Kurtz, et al. 2016), though research is still ongoing. Electricity is potentially even more difficult to evaluate due to the range of possible sources. For example, switching to electric vehicles will eliminate pollution from the point of use, but may have a net positive or negative impact on overall air quality, depending on the source of electricity generation (Michalek et al. 2011).

Overall, there is much variability in sustainability impacts of fuel production as no two fuel production (and use) pathways are identical, even for the same fuel produced from the same feedstock (Orellana et al. 2018). For any fuel it is critical to have up to date information on the entire life cycle of its production and use, to evaluate, monitor and manage potential sustainability impacts.

Incorporating sustainability criteria (like the environmental considerations listed above) within an LCFS framework is challenging. While there is considerable precedence for quantifying some of these non-GHG environmental burdens in LCAs, this approach alone is often insufficient for assessing their true impact. For example, an LCA framework can capture key burdens such as criteria air contaminants, water use, and releases to water. The importance of these burdens, however, depends on additional information such as: how close to population centres are the air pollutants released? Is water use occurring in regions with high water scarcity/stress? How sensitive are the aquatic environments into which water releases occur? Other impacts, such as biodiversity, may be even more difficult to address due to the current state of the science. While there exist some metrics for measuring biodiversity, the impacts are difficult to quantify and rarely included in LCAs (Reap et al. 2008). Further, some sustainability metrics require more contextual information about how exactly the fuel is being used. For example, the types and quantities of air pollutant emissions will depend on engine/boiler design and control technology as well as the fuel source employed. Thus, while conceptually easy to capture in an LCA framework, they would be exceptionally difficult to regulate within the context of an LCFS policy alone. Finally, incorporating sustainability criteria within an LCFS framework poses a particular challenge as fuels are expected to be regulated by a single value (i.e., carbon intensity), which does not lend itself to the multi-dimensional framework that sustainability demands. Thus, while non-GHG aspects are often noted in existing LCFS-type regulations, these regulations remain GHG emission policies at their core.

As described in Table 1, both the US Renewable Fuel Standard and the EU Fuel Quality Directive address sustainability criteria through simple limitations to the types of fuel feedstocks that qualify – primarily by restricting biofuels from new land, sensitive land and/or food crops. During the development of the California LCFS, there was investigation and consideration of the inclusion of environmental sustainability aspects but ultimately it was determined that these concerns were best addressed by other regulations independent from the LCFS.

4.1. SUSTAINABILITY CRITERIA: RECOMMENDATIONS

At a minimum, we recommend a LCFS regulator should plan periodic reviews of the non-GHG sustainability impacts of the policy. The review should include, but not necessarily be limited to categories such as those listed above in Section 4. Although it may not be possible to incorporate non-GHG environmental sustainability criteria directly within the LCFS carbon intensity framework, there are nevertheless several steps that policy makers can take to minimize the risk of adverse outcomes. Such steps may include:

- Reviewing and adjusting or adopting complementary policies (e.g., vehicle emission regulations) to ensure the most harmful impacts are appropriately managed outside the scope of the LCFS policy
- Placing relevant restrictions on which fuels can qualify for LCFS credits; for example, by excluding energy sources grown/extracted from land with high carbon stocks (e.g., as in the EU Fuel Quality Directive).
- Creating additional incentives or penalties (e.g., by adjusting the number or value of credits created) for fuels with known low/high non-GHG sustainability burdens (e.g., incentivizing the use of "waste" feedstocks). See also recommendation in section 3.6 regarding land use intensity and promotion of energy sources from marginal/degraded land or using practices with low risk for induced land use change.

Regardless of the approach taken, policy makers should recognize that carbon intensity alone is not sufficient to determine the desirability of different fuel choices. They should therefore be willing to make adjustments to ensure appropriate fuels are being incentivized.

5 | SUMMARY OF KEY POINTS FOR DEVELOPMENT AND REFINEMENT OF LOW-CARBON FUEL STANDARD POLICIES

The monitoring of life cycle GHG emissions intensity of fuels, and the creation of incentives to reduce these intensities, is an opportunity to achieve substantial reductions in GHG emissions in the transportation sector, industry, and built environment. However, it is critical that LCFS frameworks are designed in a manner that ensures fuels are being properly incentivized based on their carbon intensity and broader sustainability attributes. The following are key considerations that warrant attention in the development and refinement of LCFS frameworks:

- Calculating life cycle carbon intensity values is not an exact science. LCFS policies should outline clear GHG accounting methodologies to ensure consistency between estimates for all fuels considered within the regulation.
- LCFS policies should make adjustments to account for differences in efficiency among fuels used for any given end-use.
- Uncertainty within carbon intensity estimates should be addressed explicitly and quantitatively within the LCFS framework.
- Indirect effects such as land use change may influence the net GHG changes brought about by LCFS policies, but are difficult to quantify or validate. Careful consideration must be given to how/if the policy will account for these effects.
- There exist a number of models and methods to account for emissions from land use change and other indirect effects; many can be adapted to other jurisdictions. Alternative strategies to address indirect effects may include fuel restrictions or additional incentives, setting minimum GHG emission reduction thresholds, or various complementary policies.
- Non-GHG environmental sustainability criteria should not be excluded from consideration, even though they may be difficult to account for within an LCFS. These can likewise be addressed through appropriate fuel restrictions and complementary policies.

Given the numerous challenges in defining an appropriate LCFS framework and associated GHG accounting methodology, it is imperative that these be deployed in a manner that will improve understanding of the important and complex issues discussed in this white paper, through the collection of data and development of analytical techniques/ models, and also that the policies are able to evolve over time to make best use of the contemporary evidence base.

Revisions

The table below details the revisions to the PDF online version of *White Paper: Global Best Practices in Low-Carbon Fuel Standards*, originally published by Pollution Probe and Savant Technical Consulting on August 31, 2018. A revised version of this paper was published on November 9, 2018.

The original report explained the need for LCA models to establish procedures for on-going updating of carbon intensity values with regularly collected data. While this point remains valid, the example provided was not an accurate characterization of current procedures in GHGenius. Although GHGenius uses 1999 as the base-year for corn ethanol production data, that value is updated with trends fit to more recent data, meaning that the final results are likely more representative of the recent past than suggested by the 1999 base-year. The section in question (2.2) has been updated in the revised report.

Location	Original Text	Revised Text
Section 2.2, paragraph 3	Approaches to data collection and interpretation in existing LCA models may not be appropriate for use in an LCFS framework. For example, GHGenius widely employs time series to estimate key data (agricultural yield; biofuel production inputs and yield), with values from a reference year extrapolated to the present and future based on assumed trends. Some base data in the most recent publicly available version (v5.0a) are more than 20 years old (as of 2018), and so it is important to verify where possible if projections are relevant for current and near-term production systems. As an example, data in GHGenius v5.0a for production inputs to corn ethanol are sourced from 1999, with assumed reductions in electricity and natural gas requirements (2% and 1.8%/yr, respectively) and improvement in ethanol yield (0.2%/yr), which leads the model to calculate correspondingly lower GHG intensity with each passing year. As such, GHGenius' assumptions for current production are considerably different from the base data (~30% reduction in process energy consumption). Establishing procedures for on-going updating of carbon intensity values with regularly collected data would provide greater accuracy than modeled trends.	Models as compliance tools typically allow fuel producers to enter primary data for key inputs to ensure results are representative of current processes. Where primary data are unavailable, model defaults can be used, although these default values require regular updates for accuracy. Default values may or may not be appropriate for use in LCFS. As an example of default values, GHGenius widely employs time series to estimate default values (agricultural feedstock yield; biofuel production inputs and yield), with values from a reference year extrapolated to the present and future based on observed historical trends. Although some base data in the most recent publicly available version (v5.0a) are about 20 years old (as of 2018), the fitted trends typically account for more recent data, allowing the model to calculate values representative of more recent years. Data in GHGenius v5.0a for production inputs to corn ethanol use 1999 as a base year, with fitted reductions in electricity and natural gas requirements (2% and 1.8%/yr, respectively) and improvement in ethanol yield (0.2%/yr), which leads the model to calculate correspondingly lower GHG intensity with each passing year. As such, GHGenius' default assumptions for current corn ethanol production are considerably different from the base data (~30% reduction in process energy consumption), which partly reflects actual historical reductions in process energy use since 1999, and partly reflects the assumption that these trends continue to the present. Modeled trends allow users the ability to forecast future performance, but still require procedures for on-going updating of carbon intensity values with regularly collected data to ensure modeled results remain current.
Section 3.3, Figure	Figure 2. Schematic representation of how economic equilibrium models (EEMs) are used	Figure 2. Schematic representation of how economic equilibrium models (EEMs) are used to

2 heading	to estimate GHG emissions from induced land use change (ILUC).	estimate GHG emissions/sequestration from induced land use change (ILUC).
Section 3.6, second bullet point	Restricting the use of fuels from certain feedstocks, such as biomass grown on virgin land or land with high carbon stocks (e.g., as in the EU Fuel Quality Directive).	Restricting the use of fuels from certain feedstocks, such as biomass grown on land with high carbon stocks (e.g., as in the EU Fuel Quality Directive).
Section 3.6, bullet point added		Promoting the development of energy sources (i) grown on/placed on/extracted from marginal or degraded lands, or (ii) grown by adopting practises considered to introduce a low risk for induced land use change (e.g., *new* implementation of cover crops or double cropping, or growth of a biofuel crop during what had normally been a fallow period). For example, the EU Fuel Quality directive provides a bonus (negative emission value) of 29 g CO2e/MJ biofuel if the feedstock is grown on previously degraded land.
Section 4, paragraph 3	There are numerous ways in which the adoption of alternative fuels may impact (positively or negatively) these other (non-GHG) sustainability categories. For example, production of liquid biofuels often consumes orders of magnitude more water than petroleum fuels, potentially exacerbating issues of water scarcity depending on where and how the feedstocks are grown (Dominguez-Faus et al. 2009; Spang et al. 2014). Without careful management, biofuels may also lead to important disruptions in the nitrogen cycle, potentially leading to water quality issues like eutrophication (Alshawaf, Douglas, and Ricciardi 2016; Miller, Landis, and Theis 2007). Any fuel that contributes to land use change, may likewise have important impacts on biodiversity due to habitat destruction. In contrast, fuels like biodiesel may exhibit lower toxicity than petroleum diesel (e.g. Khan, Warith, and Luk 2007). Existing literature suggests that increased use of biofuels like ethanol and biodiesel has little adverse impact (and possibly some benefits) for air quality (NOx, non-methane hydrocarbons, particulate matter and mobile source air toxics) relative to fossil fuels (Wallington, Anderson, Kurtz, et al. 2016), though research is still ongoing. Electricity is potentially even more difficult to evaluate due to the range of possible sources. For example, switching to electric vehicles will eliminate pollution from the point of use, but may have a net positive or negative impact on overall air quality, depending on the source of electricity generation (Michalek et al. 2011).	There are numerous ways in which the adoption of alternative fuels may impact (positively or negatively) these other (non-GHG) sustainability categories. Potential impacts could transcend various fuel types, from renewables to fossil fuels, and some feedstock/fuel pathways will have lower/ higher sustainability impacts than others. For example, some authors have raised concerns related to water consumption and water quality associated with fuel production, although the extent and magnitude of such impacts are not well understood. Recent work commenced at the Argonne National Laboratory and the US Department of Energy's Biomass Energy Technologies Office (BETO) aims to develop quantitative metrics based on water consumption analysis and hydrologic models, and evaluate sustainable water use scenarios in the production of bioenergy and bioproducts (Wu, Ha, and Xu 2017) Natural Resources Canada has recently developed I-BIOREF (Natural Resources Canada 2015), which enables comparison of biorefinery configurations and includes water withdrawal as well as other sustainability metrics. From a process perspective, water consumption attributable to biofuels production is relatively low, depending upon the process – and reported to be roughly on par with water use attributable to petroleum fuel production according to an international study (Spang et al. 2014). More substantive impacts have been claimed if the feedstock requires irrigation. For example, in 2011, the developers of GREET calculated average water consumption for corn ethanol to range from 11-160 gallons of water per gallon of ethanol, corresponding to U.S. corn irrigation rates ranging from 3-35% of harvested crop, compared with an average of 2.6-6.6 gallons of water per gallon of petroleum gasoline, which encompasses their

		estimates for both conventional and oil sands production (Wu and Chiu 2011). This example is specific to corn ethanol in the United States, and does not reflect on all biofuels/feedstocks. In contrast, irrigation is currently relatively uncommon in Canada. Some crops are irrigated in Southern Alberta, Southern Saskatchewan, and Southern Ontario, yet in 2017, in Alberta only 4% of all wheat, and only 2.6% of all canola, was irrigated (Alberta Government 2017). In Ontario, only 2% of all field crops were irrigated in 2016 (Statistics Canada 2018b)(Statistics Canada 2018a). While the current impact of bioenergy on water use in Canada may be relatively small, it is important to be aware of and manage potential impacts should circumstances change. The BETO work cited above, for example, has identified various practises that can be adopted to manage water consumption and discharge, particularly from feedstock production. Similarly, it is important to be aware of and manage other sustainability impacts of fuels, from impacts on groundwater to water quality issues like eutrophication (Alshawaf, Douglas, and Ricciardi 2016; Miller, Landis, and Theis 2007). Awareness of these potential issues becomes key to their management, such as through improved methods for wastewater treatment, those to deliver nitrogen to crops or the use of precision agriculture. Fuels (conventional and alternative) that lead to changes in land use and habitat destruction may also impact biodiversity. It is therefore important to monitor and manage such potential impacts. Issues like fuel toxicity and impacts of fuels on air quality are also critical sustainability aspects. For example, fuels like biodiesel may exhibit lower toxicity than petroleum diesel (e.g. Khan, Warith, and Luk 2007). As another example, existing literature suggests that increased use of biofuels like ethanol and biodiesel has potential benefits for air quality (NOx, non-methane hydrocarbons, particulate matter and mobile source air toxics) and should not result in increased emissions when used in modern gasoline and diesel vehicles and when biofuel content is within the operational ranges for which the vehicles are designed (Wallington, Anderson, Kurtz, et al. 2016), though research is still ongoing. Electricity is potentially even more difficult to evaluate due to the range of possible sources. For example, switching to electric vehicles will eliminate pollution from the point of use, but may have a net positive or negative impact on overall air quality, depending on the source of electricity generation (Michalek et al. 2011). Overall, there is much variability in sustainability impacts of fuel production as no two fuel production (and use) pathways are identical, even for the same fuel produced from the same

		feedstock (Orellana et al. 2018). For any fuel it is critical to have up to date information on the entire life cycle of its production and use, to evaluate, monitor and manage potential sustainability impacts.
Section 4.1, second bullet point	Placing relevant restrictions on which fuels can qualify for LCFS credits; for example, by excluding energy sources grown/extracted from sensitive or undeveloped lands and/or limiting the use of food crops for biofuel production (as in the EU Fuel Quality Directive).	Placing relevant restrictions on which fuels can qualify for LCFS credits; for example, by excluding energy sources grown/extracted from land with high carbon stocks (e.g., as in the EU Fuel Quality Directive).
Section 4.1, third bullet point	Creating additional incentives or penalties (e.g., by adjusting the number or value of credits created) for fuels with known low/high non-GHG sustainability burdens (e.g., incentivizing the use of "waste" feedstocks)	Creating additional incentives or penalties (e.g., by adjusting the number or value of credits created) for fuels with known low/high non-GHG sustainability burdens (e.g., incentivizing the use of "waste" feedstocks). See also recommendation in section 3.6 regarding land use intensity and promotion of energy sources from marginal/degraded land or using practices with low risk for induced land use change.
Additions to References		Alberta Government. 2017. *Alberta Irrigation Information.* Natural Resources Canada. 2015. *I-BIOREF Software. Modeling and Analysis for Technical, Economic, and Environmental Assessment of Biorefinery Processes.* Orellana, A., I.J. Laurenzi, H.L. Maclean, and J.A. Bergerson. 2018. "Statistically Enhanced Model of in Situ Oil Sands Extraction Operations: An Evaluation of Variability in Greenhouse Gas Emissions." *Environmental Science and Technology* 52(3): 947–54. Statistics Canada. 2018a. "Hay and Field Crops. Table: 32-10-0416-01." ———. 2018b. "Total Area That Received Irrigation by Crop Type. Table: 38-10-0241-01." Wu, M., and Y. Chiu. 2011. *Consumptive Water Use in the Production of Ethanol and Petroleum Gasoline-2011 Update.* Wu, M., M. Ha, and H. Xu. 2017. *Impact of Projected Biofuel Production on Water Use and Water Quality.*
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APPENDIX I

INDUCED LAND USE CHANGE

Induced land use change (ILUC)[6] of biofuel production: Additional background, current modeling efforts and policy formulations

1. Land use intensity of different energy systems
2. Major approaches for modeling induced land use change
3. Characteristics of prominent economic equilibrium models for ILUC modeling
4. Emission factor models used in land use change modeling
5. Approaches of current policies to address induced land use change

[6] Acronyms for land use change are used inconsistently in the literature. 'ILUC' has been used to represent the international component of indirect land use change, all indirect land use change, or all induced land use change (including direct and indirect). This comment uses the latter, broadest definition of induced land use change.

1. LITERATURE REPORTED LAND USE INTENSITY OF DIFFERENT ENERGY SYSTEMS

Table A1 presents estimates for the land use intensity of different energy sources as reported in the literature. Note that this data describes how much land is typically required to produce a given quantity of energy, but does not estimate land use change induced by these energy sources.

TABLE A1. LAND USE INTENSITY OF DIFFERENT ENERGY SYSTEMS AS REPORTED IN LITERATURE

			Land use intensity (m2/GJ)		
Product	**Energy system**		**U.S [1]**	**Germany [2]**	**Canada [3] [4]**
Electricity	Nuclear		0.03 (0.01-0.06)	0.28	NR
	Natural gas	Conventional	0.28 (0.22-0.28)	0.03	NR
		Tight Gas	0.06 (0.03-0.25)	NR	NR
	Coal	Underground	0.17 (0.06-0.42)	0.06	NR
		Surface	2.28 (1.31-4.56)	0.11	NR
	Renewables	Wind*	0.36 (0.08-0.39)	0.19	NR
		Geothermal	1.42 (0.58-3.06)	0.69	NR
		Hydropower	4.69 (1.81-24.17)	0.97	1.03
		Solar photovoltaic	4.17 (3.42-4.72)	2.42	NR
		Solar- thermal	5.36 (3.61-7.78)	2.17	NR
		Biomass	225 (155-348)	125	NR
Liquid Fuel	Fossil oil	Conventional	0.17 (0.14-0.19)	0.03	0.028 (0.014-0.056)
		Tight Oil	0.11 (0.06-0.25)	NR	NR
		Oil sands- surface mining	NR	NR	0.011(0.008-0.017)
		Oil sands- in situ	NR	NR	0.003 (0.003-0.006)
	Biofuels	Corn	66 (54-72)	61	NR
		Sugarcane	76 (63-95)	66	NR
		Soybean	82 (66-87)	133	NR
		Cellulose	156 (35-229)	114	NR
		Cellulose residue	NR	0.03	NR

Values are single estimates or the mid-range values and the lower-bound and upper-bound estimates are reported in the parentheses. Differences in land use intensity for the same energy system that use the same resource are due to local circumstances (e.g., spacing of oil and gas wells, wind regime, level of insolation, dam height, biomass yields). Specific fuels (e.g., the type of fossil or biofuel is assumed to be produced) are not noted in the studies and therefore not included in the table.

*Excludes land between structure elements. The land around can fulfill other functions such as agriculture. NR = not reported.

2. MAJOR APPROACHES FOR MODELING INDUCED LAND USE CHANGE

As Induced Land Use Change (ILUC) stems from economy-wide interactions it must be modeled rather than measured. Models used to predict the quantity and location of ILUC generally fall into three general categories: causal descriptive models (CDMs); economic equilibrium models (EEMs); and integrated assessment models (IAMs).

CDMs use cause and effect logic to describe and derive the ILUC impacts. The market responses are estimated using a combination of historical trends, input and validation of future markets by experts and stakeholder feedback [5][6]. In principle, the advantages of CDMs relate to a more participative approach of stakeholders and also that they allow accounting for non-economic LUC drivers such as political and technological trends. However, the results of the CDMs are critically dependent on the accuracy of the assumptions of experts and on the future conditions extrapolated from historical trends, which are highly uncertain. In addition, market mechanisms are simplified and not fully considered in CDMs [7].

EEMs are models that assume perfect markets that reach an equilibrium wherein demand equals supply in the studied economy. EEMs start with a baseline supply and demand equilibrium across the world economy or within a set of specified interconnected markets. An economic shock is introduced (e.g., increase in demand of biofuel due to a new policy), and the model forecasts the resulting new equilibrium. EEMs rely on estimated elasticities in market supply/demand, which are estimated from historical data. For biofuel analysis, standard EEMs are generally adapted for the application. Commonly EEMs predict net changes in land use, and are then coupled with an external model to calculate the greenhouse gas (GHG) emissions resulting due to these changes.

IAMs are models that describe the interactions between human activities and environmental change processes (e.g., land use, GHG concentration, temperature change). They connect a broad range of domains into a single framework. Example domains may include economics, biogeochemistry, engineering/technology, and atmospheric science. IAMs have been used to project large-scale use of agricultural and forest residues and herbaceous energy crops. In contrast, they have less often been used to examine first generation bioenergy crops (i.e., food crops, like corn grain) due to their focus on a long-time horizon. The strength of IAMs is that synergies and feedbacks of policy strategies can be assessed in different domains. However, these models are very complex and rely on numerous assumptions [8,9]. IAMs that have analyzed ILUC from bioenergy production include the IMAGE (Integrated Model to Assess the Global Environment) framework [10] and the Global Change Assessment Model (GCAM) [11] [12]. This latter model, for example, is a long-term (operating over a projected time horizon from today through 2095 and operates in 5-year time steps), integrated model with the global economy, energy system, agriculture and land use, and emissions included in its coverage. The model's economic behavior is based on the concept of a recursive, dynamic market equilibrium; it adjusts prices of all energy, agriculture, and forest products until supplies and demands of each reach an equilibrium. IAMs have not yet been used in a regulatory context for ILUC assessment.

Existing regulations have relied primarily on EEMs for their analysis of ILUC. EEMs are currently used in the calculation of carbon intensity values in the US policies (California LCFS, US Renewable Fuel Standard) and to evaluate ILUC in Europe (EU Fuel Quality Directive). Since there are a large number of models and studies that consider ILUC, the scope of the reviewed models in the next sections is limited to the prominent EEMs currently used in biofuel policies.

3. CHARACTERISTICS OF PROMINENT ECONOMIC EQUILIBRIUM MODELS (EEMS) FOR ILUC MODELING

Within EEMs for biofuel-induced ILUC, there are General Equilibrium Models (GEMs) and Partial Equilibrium Models (PEMs). GEMs include demand functions and production functions (i.e., an abstraction of how capital, labor, and other broad input categories are combined to create representative final products) for an aggregated set of sectors that represent the entire economy. PEMs provide supply and demand functions that focus on specific sectors of interest for ILUC. Prominent GEMs include the Global Trade and Analysis Project with Biofuels (GTAP-BIO) [13] and Modeling International Relationships in Applied General Equilibrium- BioFuel (MIRAGE-BioF) [14]. Prominent PEMs include the US-focused Forest and Agricultural Sector Optimization Model (FASOM) [15], the Food and Agricultural Policy Research Institute (FAPRI) model [16], and the European-focused Global Biosphere Management Model- European Union (GLOBIOM-EU) [17]. Several of these models are used in existing LCFS regulations (Tables 1 and A2). To date, no EEM model has been developed to specifically assess ILUC impacts of Canadian biofuel policy; however, existing tools could be adapted to do so.

The major characteristics of the prominent EEMs currently used in biofuel policies are listed in Table A2. Note that these models are particular versions of the standard models that were specifically refined and or adapted for analyzing biofuel policies (e.g., GTAP-BIO). In this section we focus on these specific biofuel versions because they have additional characteristics for biofuel analysis compared to the standard models that were designed for other and/or broader purposes. However, the structure and main characteristics of the biofuel version and standard model are the same.

GTAP-BIO and MIRAGE-BioF are two GEMs that share similar characteristics:

- They are based on a top-down approach starting from macroeconomic accounts.
- They model all sectors of the economy, though in an aggregated manner.
- The substitutions between local and foreign goods rely on Armington trade elasticities, which means that products traded internationally are differentiated by country of origin (i.e., equivalent goods from different countries are treated as imperfect substitutes, meaning that importers can continue to source goods from multiple countries even if there are price differentials among them).
- Competition for land use relies on a constant elasticity of transformation (CET), which specifies how much land is transformed (in percent) to a given category in response to a percentage increase in the rent (i.e., relative value) of that particular land use. The model will first apply a CET function to determine how much land is devoted to high level categories like cropland and grassland, and will subsequently apply another CET function to select among crop types within the cropland category.
- They rely on the fully documented publicly available global GTAP database. This market database is mainly composed of regional social accounting matrices that capture economic flows between economic agents in value terms.

Partial Equilibrium Models (PEMs), such as (FASOM), (FAPRI) and (GLOBIOM-EU), also share similar characteristics:

- They provide high resolution within the agricultural sector, but do not capture linkages to other sectors of the economy.
- Goods are considered homogeneous and are not differentiated by national boundaries; hence the lowest cost provider will always be used.
- Competition for land use is modeled using data on land characteristics together with data on yields and costs to allocate different crops explicitly to specific units of land, so as to maximize economic returns. Modeling expansion of overall land availability is possible, but has historically been challenging for PEMs [18].
- According to the region of the study, data are extracted from different sources: for example EUROSTAT for Europe and USDA National Agricultural Statistics Service for the U.S.

Overall, GEMs provide a broader perspective regarding the different market interactions and assess the economy-wide feedback effects (e.g., an increase in oil price will translate into an increase in production of a biofuel and thus its demand for an agricultural feedstock) [19]. However, GEMs do not capture all important characteristics of the agricultural economy, relying instead on a set of broad and highly simplified/aggregated characteristics [20]. Conversely, PEMs often provide greater resolution for changes within agriculture and forestry sectors, but are unable to assess induced changes in other economic sectors.

Other key differences between the EEMs are summarized in Table A2, including:

- Geographic coverage (e.g., regional vs global) – particularly important with regard to relevance for countries like Canada that may not be included in regional models.
- Whether the model is static (i.e., simply compares economic equilibria in hypothetical worlds with and without the ‘policy shock’) or dynamic (e.g., projects economic and land use changes for specific future years).

- Spatial resolution, which can vary dramatically from 19 regions with 18 agro-ecological zones (AEZs) up to tens of thousands of individual land units.
- The way in which the model interacts with emission factor databases to calculate GHG emissions (e.g., which database? Is it embedded in the model or does the model need to be coupled with an external database). This is elaborated in section 3.

Table A2 provides only a high level overview of model differences. There exist many subtle modeling decisions that can have important impacts on model results. One prominent example relates to how co-products are treated. Biofuel production yields valuable protein-rich co-products that can be used as animal feed (e.g., dried distillers' grains and solubles (DDGS), canola and soybean meals). Some models, like GTAP-BIO, FASOM and FAPRI assume co-production of feed (e.g., DDGS) will displace existing feed crops, thereby reducing net land use. Others, like GLOBIOM-EU, allow some co-produced feed instead to drive the expansion of livestock production. Subtle structural differences like this can profoundly impact the amount of land use change predicted by the models.

TABLE A2. MAJOR CHARACTERISTICS OF ECONOMIC EQUILIBRIUM MODELS (EEMS) FOR LAND USE CHANGE

Models	GTAP-BIO[a]	MIRAGE-BioF[b]	FASOM[c]	FAPRI[d]	GLOBIOM-EU[e]
Policies	California's LCFS	European RED	EPA RFS2 US LUC	EPA RFS2 Outside US LUC	European RED
Type of EEM and economic sectors	General equilibrium; 57 economic sectors	General equilibrium; 55 economic sectors	Partial Equilibrium; Agriculture, Forestry	Partial Equilibrium; Agriculture, Biofuels, Livestock, Dairy	Partial Equilibrium; Agricultural, livestock, forestry and bioenergy sectors
Model framework	Top-down, starts from macroeconomic accounts	Top-down, starts from macroeconomic accounts	Bottom-up, starts from land and technology	Behavioral equations for production, consumption, and trade. Change in price triggers response in acres grown	Bottom-up: starts with land cover, land use, management systems to extrapolate supply
Geographic Coverage and land resolution	Global World by AEZ 19 regions (geographic) with 18 AEZ types	Global World by AEZ 18 AEZ	US by county	World by political borders	Global (28 EU Member states + 29 regions) Detailed grid-cell level (>10,000 units worldwide)
Trade assumptions	Armington Elasticities	Armington Elasticities	Homogeneous Goods	Homogeneous Goods	Homogeneous Goods
Market data source	GTAP harmonized with FAOSTAT	GTAP harmonized with FAOSTAT	USDA	FAOSTAT, USDA	EUROSTAT and FAOSTAT
Static or Dynamic	Typically static, but dynamic version exists	Dynamic	Dynamic	Dynamic	Dynamic
Time Frame	2011 Database with elasticities calibrated for medium term (5 years or 15-20 years)	2004 Database with elasticities calibrated for 2020 timeframe (RED) (one year time step)	Projects up to 100 years with intervals every 5 years	10-15 years	2000-2030 (ten year time step)

Models	GTAP-BIO[a]	MIRAGE-BioF[b]	FASOM[c]	FAPRI[d]	GLOBIOM-EU[e]
Land cover types	Cropland, forest, pasture, cropland-pasture included for US and Brazil	Arable land, meadows and permanent pasture, permanent crops, cropland, managed forest	Cropland, cropland-pasture, forest-pasture, rangeland, forest, developed, conservation reserve program	Cropland, forest, pasture, barren	Cropland, other agricultural land, grassland, forest, wetlands, other natural land
Model Outputs	Amount of land and change in land cover type (e.g. pasture to crop) by AEZ	Amount of land and change in land cover type (e.g. pasture to crop) by AEZ	Emissions of land use + crop GHG changes + livestock GHG changes in g/MJ for the entire US + fertilizer use	Amount of land needed by crop and geographic region	Emissions of land use + crop GHG changes+ livestock GHG changes+ fertilizer use + manure + soil carbon
Carbon accounting (external or endogenous to the model)	External: AEZ-EF model IPCC GHG inventories and default values, but utilizes more recent data when available Note that other carbon accounting models have been used like Wood Holes	Endogenous managed LUC combined with marginal land cover type change from MODIS Satellite Data combined with WINROCK carbon stock factors	Endogenous: CENTURY Models net CO_2, N_2O and the CH_4 endogenous to the model due to estimated LUC Linked to IPCC, DAYCENT and FORCARB emission factor databases	External: LUC combined with either (1) marginal land cover type change from MODIS Satellite Data combined with WINROCK carbon stock factors or (2) IPCC parameters	Endogenous: 12 sources of GHG emissions covering crop cultivation, livestock, land use change, soil organic carbon based on advanced accounting framework. Peatland IPCC emissions values revised upward based on exhaustive recent literature review.

Notes: AEZ= agro-ecological zones (AEZs); EF= emission factor
[a] Sources: [21][22][13][23]
[b] Sources: [22][24][14]
[c] Sources: [22][15]
[d] Sources: [22]
[e] Sources: [25] [17]

4. EMISSION FACTOR MODELS USED IN LAND USE CHANGE MODELING

Although reviews of ILUC modeling tend to focus on how models predict which land changes will occur, the carbon stock databases and emission models that link land transformations to GHG emissions are an equally important consideration. There is substantial uncertainty in estimating changes in carbon stocks (particularly soil carbon), along with heterogeneity regarding the extent to which the models account for non-CO_2 GHG emissions arising from ILUC (e.g., N_2O emissions associated with fertilizer use).

The major categories of ILUC emissions that have been most examined are those related to CO_2 emissions due to changes in vegetative carbon stock, soil carbon stock and aboveground forgone carbon sequestration. Vegetation and soil carbon databases developed by the Woods Hole Research Center (WHRC), Winrock International (WI), Intergovernmental Panel on Climate Change (IPCC), and the California Air Resources Board (AEZ-EF) have provided the basis for many research studies to estimate ILUC emission factors. These databases are generally based on empirical models that consist of regression analysis to extrapolate existing research and data to develop regionally explicit emissions factors relevant to different land uses and land management. This approach is relatively easy and transparent to use, but is also highly aggregated and thus cannot consider location-specific soil/weather parameters or influences of past management. For example, the AEZ-EF model (designed specifically to work at the same resolution as GTAP) allocates global land cover into 19 distinct regions and 18 distinct agro-ecological zones; within each, a single GHG emission value is considered representative of each type of land cover or management change.

In contrast, process-based biogeochemical models like CENTURY and DAYCENT use mechanistic equations based on substantial long-term research to represent growth, nutrient, water, soil, and GHG dynamics. As a result, these models can provide more detailed modeling and are well matched for the high resolution of PEMs with respect to modeled land transformations and land management practices. This resolution comes with increased complexity, however, requiring significantly more site-level data inputs and detailed verification.

5. APPROACHES OF CURRENT POLICIES TO ADDRESS INDUCED LAND USE CHANGE

5.1 CALIFORNIA AIR RESOURCES BOARD (CARB)

CARB determined that land use change induced by biofuel demand is important and must be included in LCFS fuel carbon intensities [26]. The GTAP-BIO economic model was deemed the best tool available to estimate GHG emissions from ILUC because of its long history in modeling complex international economic effects and its global scope [27]. Given uncertainty and variability associated with parameters that have an important influence on the model outputs, CARB used a scenarios approach. The output ILUC GHG emissions from all scenarios were averaged and represent the carbon intensity value for ILUC for a specific crop-derived biofuel. In addition CARB performed an uncertainty analysis based on Monte Carlo simulation to derive output probability distributions for ILUC GHG emissions [26]. CARB uses a 30-year timeframe for calculating ILUC values.

5.2 OREGON DEPARTMENT OF ENVIRONMENTAL QUALITY (DEQ)

For the Oregon Clean Fuels Program DEQ has taken the ILUC factors directly from the CARB LCFS re-adopted regulation of 2015, except the value for corn ethanol. DEQ used the corn ILUC factor estimated by the GTAP model coupled with the CCLUB emission model of Argonne National Laboratory, which uses CENTURY to model changes in soil carbon, together with either Winrock or Woods Hole databases for domestic and international carbon stocks and/or N_2O emissions [28]. Like CARB, DEQ uses a 30-year timeframe for calculating ILUC values.

5.3 US ENVIRONMENTAL PROTECTION AGENCY (EPA)

The EPA developed an approach for RFS2 linking FASOM with CENTURY to estimate domestic LUC and FAPRI coupled with Winrock International land use change emission factors to estimate International LUC GHG emissions [19]. The total LUC factor is the sum of US land use change and non US land use change for a specific crop. This value is included in the total GHG emissions estimate for a specific biofuel in the RFS2 [29]. The two models are independent but the FASOM outputs are used as inputs to FAPRI and they reach equilibrium separately. As they are not interconnected, there could be some mismatches at the end of the model runs such as the values of U.S. exports and import quantities and agricultural prices [30]. For the final rule some modifications were performed such as the incorporation of a forestry model in FASOM, revision of co-product substitution rates, addition of corn oil as a co-product, and the addition of a Brazilian agriculture model. Compared to the other policies above, the EPA approach is arguably moving closer to a consequential LCA in that it includes factors like N_2O emissions from fertilizer application as a component of its overall land use modeling, rather than as part of the supply chain for individual biofuels as is done for the other policies.

EPA used a 30-year timeframe in the calculation of ILUC factors. They also explored the possibility of applying a discount rate (akin to an economic discount rate) to capture the temporal profile of these ILUC emissions, but eventually decided against the approach.

5.4 EUROPEAN COMMISSION

The European Commission acknowledged that ILUC can increase GHG emissions during biofuel production. However, given the uncertainty linked to model-based estimates of ILUC GHG emissions of biofuels, the EU recommended addressing this issue under a precautionary principle approach and defined sustainability criteria that biofuels must satisfy [31]. At present, ILUC emissions are reported in the RED and the FQD, but are not accounted for in the sustainability criteria or in regulatory GHG emissions accounting.

Several ILUC studies have been commissioned by the European Commission (e.g., AGLINK, MIRAGE-BioF, CAPRI, LEITAP, GTAP Europe, IMPACT, GLOBIOM-EU) [32][24]. The last study commissioned was based on a partial equilibrium model, GLOBIOM-EU [25]. The European Commission acknowledge that the results of the studies vary for the same feedstock, mainly due to the complexity and variety of the models used for estimating the ILUC factors [31]. Therefore, the European Commission decided that more research on land use change emissions was needed. Notably, European ILUC studies use a timeframe of 20 years consistent with IPCC accounting but not consistent with those of other jurisdictions.

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APPENDIX II

SUSTAINABILITY CRITERIA

Description of common environmental sustainability categories

- Climate change: the release of greenhouse gases (GHG) like CO_2, N_2O and CH_4 can trap outgoing infrared radiation, upsetting the global heat balance and leading to changes in the climate that can worsen sea-level rise, extreme weather, wildfires, and more. This is the key focus for LCFS policies.
- Non-renewable resource depletion: often separated into subcategories, such as the consumption of fossil fuels, dispersion of metals or other minerals, this category encompasses a suite of concerns regarding the use of resources which cannot be regenerated on a human time-scale.
- Water withdrawals and consumption: water use is often differentiated based on whether the water is withdrawn then returned to its source (as is common for cooling water) or consumed (e.g., evaporative losses, used in agricultural irrigation). Water consumption is especially problematic when it reduces the availability of freshwater, especially in water stressed regions. Water withdrawals can also create environmental issues, for example if the water is returned with contaminants or at a higher temperature, potentially creating inhospitable environments for aquatic life.
- Acidification: the process that increases the acidity of water and soil, changing local ecosystems that can result in the death of plants and animals. Emissions of gases like nitrogen oxides (NOx) and sulfur dioxide (SO2) are common contributors to acidification.
- Eutrophication: the process whereby anthropogenic nutrients such as nitrogen and phosphorus runoff into natural water systems, leading to overgrowth of aquatic plant life. Common consequences include unpleasant odor and taste, production of toxic chemicals, and death of marine life due to depletion of aquatic oxygen.
- Ozone depletion: release of gases such as chlorofluorocarbons (CFCs), which react to remove ozone in the upper atmosphere, reducing the protection the ozone layer would otherwise provide against UV radiation which can damage crops, human skin and eyes, etc.
- Photochemical oxidation: release of volatile organic compounds and NOx can induce photochemical oxidation reactions that lead to the creation of ground-level ozone. Ozone is a key component of smog, which is associated with negative impacts on plant life as well as increased human mortality and respiratory conditions.
- Criteria air pollutants: in addition to smog, other air contaminants, such as particulate matter, NOx and SO2 are also associated with a range of human health complications including respiratory disease and premature mortality.
- Other negative human health impacts: a wide range of other pollutants can negatively impact human health, including cancer-causing agents like benzene, neurotoxins like lead and methanol, and others. In many cases, the full range of human health impacts from different substances is not fully understood.
- Ecotoxicity: often delineated by whether the affected species are aquatic or terrestrial, this category captures all substances that produce toxic effects in non-human species. Ecotoxicity is a broad category, which often suffers from incomplete characterization of how substances impact different species.
- Natural land transformation and/or occupation: treated respectively as a problem of resource use (i.e., land availability) or for its secondary impacts (e.g., habitat destruction, loss of ecosystem services like carbon storage, water purification and flood control), land transformations can create numerous environmental concerns.
- Biodiversity loss: represents increases in the rate of local or global species extinctions, potentially driven by other impact categories like natural land transformation and ecotoxicity. Often, special attention is paid to local reductions in the population of at-risk species.