Lessons from the ILUC Phenomenon

Michael O'Hare and Richard J. Plevin

Abstract The impact of greenhouse gas emissions on climate change occurs both through direct life cycle emissions and direct land use change as well as through indirect land use change (ILUC). The latter, in particulars are uncertain and front-loaded: land conversion leads to a large initial discharge that is paid back through reduced direct carbon intensity relative to fossil fuels in the future. This chapter discusses approaches to make policy decisions about accounting for ILUC effects in the presence of uncertainty about the magnitude of the effect and the need to balance a precautionary desire to delay investment till the uncertainty is resolved with the cost of delaying a switch from fossil fuels to biofuels. Given the temporal variation in the trajectory of emissions, policymakers should consider using metrics other than the cumulative discharges to capture the impact of emissions on the climate and the time profile of that impact and costs of positive and negative errors in incorporating ILUC effects in policy implementation. It is also important to recognize the presence of other market-mediated effects such as the fuel rebound effect that can also offset some of the direct savings in carbon emissions from switching to biofuels.

Keywords Biofuels · Indirect land use change (ILUC) · Uncertainty · Life cycle assessment (LCA) · Climate policy

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1 Introduction

Among the most debated issues raised by biofuels is greenhouse gas (GHG) discharge caused when forest and pasture are cleared and cultivated in response to the demand for land to grow bioenergy crops. Recognition of this discharge, commonly called ILUC [for *indirect land use change* (emissions)], in several fuel regulations has engendered a lively debate over its actual size for various biofuels, and methodological progress in estimating it. It has also led to insights about how ILUC emissions, and other consequences, should be recognized and incorporated in policy design and implementation. These insights, which have implications beyond biofuels and even energy policy, are among the most interesting unanticipated consequences of studying the ILUC phenomenon. The present essay describes them and draws analytic implications that challenge some aspects of conventional energy policy analysis.

We do not discuss ILUC estimation methodology, which is well covered elsewhere in this volume, nor the size of ILUC for any fuels. Our discussion begins with the recognition that an increase in crop-based biofuel production induces a GHG discharge from land use change, (approximately) at the beginning of that production, whose size, though uncertain, is potentially large enough to affect societal preferences across competing fuels (Plevin et al. 2010). Uncertainty about the magnitude of ILUC emissions, across and within different models that estimate it, is small enough not to put biofuels completely off the table but large enough to demand attention to decision-analytic methods and explicit attention to the goals of fuel policy (Plevin et al., 2015). In addition, the distinctive time profile of ILUC emissions, nearly all at the beginning of biofuel production, makes conventional metrics of fuel climate effects like total GHG discharge intrinsically ambiguous and problematic. For example, when ILUC occurs, substituting energy-equivalent amounts of a biofuel for a fossil fuel can reduce total GHG discharge but still increase GHG-induced global warming for a long period (O'Hare et al. 2009).

2 Implicit Biofuels Policy Assumptions Have Been Challenged by ILUC

Policies promoting biofuels were initially motivated by a variety of objectives, including (in Brazil and the US especially) national independence from imported fossil fuels, and the desire of agribusiness players to expand their markets. To further support existing subsidy and quantity mandates, advocates of increased biofuel use seized on growing concern about climate change to point out that as biofuels carbon is captured from the air, their use (at least in concept) presented an opportunity to capture solar energy in a closed cycle that does not increase atmospheric GHG. Note that the present discussion applies to crop-based biofuels whose feedstocks (including some cellulosic feedstocks) compete with food, feed and fiber for arable land, not biofuels from wastes or residues; these generally do not have ILUC-like discharges and require a different kind of analysis.

The carbon directly released from burning a fuel is not the only climate effect of interest, as shown by life cycle analyses (LCAs) that count the GHGs emitted in growing, transporting, and converting the biomass (e.g., by fermentation and distillation) to a liquid fuel. Some studies have shown that these emissions greatly diminish or actually vitiate the climate benefits of, in particular, a corn ethanol system (Crutzen et al. 2007; Plevin 2009; USEPA 2010; Reay et al. 2012). Attention to this issue also highlighted large differences among the climate "greenness" and net energy content of biofuels made using different farm and refinery technologies. Debating the comparative carbon intensity of fuels at this period concealed three important assumptions, as discussed below.

2.1 GHG Intensity as a Property of a Substance

Underlying the early discussion of biofuels, and the design of biofuels policies like the California Low Carbon Fuel Standard and the US Renewable Fuel Standard was an implicit assumption that the differential climate effects of producing different fuels could be captured, well enough for policy making, by calculating each fuel's *global warming intensity* (GWI), using attributional LCA (ALCA). This is usually measured in g CO_2e MJ^{-1}, representing the "GHG [scaled to equivalent amounts of CO_2 using the IPCC's GWP methodology (Ciais et al. 2013)(IPCC)]" released by (i) making and using an amount of fuel providing one megajoule of energy, and (ii) only doing that. ALCA is a bookkeeping exercise tracing all the inputs to the final product back to their energy use, based on physical input–output relationships. The accounting stops when quantities are deemed small enough to ignore.[1] For example, the diesel fuel used in corn production is counted, but the analysis does not extend to the effect on livestock industry and its GHG discharges from either raising the price of corn or diverting it from feed. More important, as an accounting of existing practice, ALCA does not measure indirect effects across the economy of a significant change in the assortment of goods made and consumed (Ekvall and Weidema 2004; Finnveden et al. 2009). We emphasize here the implicit idea that GWI is a measure of climate effect that can be appropriately *attributed to a fuel* (perhaps a particular batch of fuel from a particular place) like its density or energy content (DeCicco 2012).

The models summarized and aligned in a widely cited 2006 meta-analysis seemed to indicate that from this perspective, corn ethanol promised modest climate and net energy benefits over fossil fuel (Farrell et al. 2006): driving a vehicle one

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[1] A small quantity of a substance can have a large environmental effect. Although cut-off criteria are typically applied in an LCA, there is no theoretical basis suggesting that the neglected elements of the life cycle are, in fact, of little significance (Plevin et al. 2013).

mile using the former would cause less global warming than driving a similar vehicle one mile on gasoline. The same paper, however, recognized that "…several key issues remain unquantified, such as…the carbon effects of conversion of forest to agriculture."

2.2 *Fuel Substitution on an Energy-Equivalent Basis*

Also implicit in the debate of the period were the assumptions that policies forcing or subsidizing biofuels into the fuel marketplace would cause replacement of fossil fuels by biofuels on a MJ-for-MJ basis, or closely enough not to matter. Some economists have questioned this MJ-for-MJ substitution rate (Drabik and de Gorter 2011; Bento and Klotz 2014) and the response of the fuel system as a whole to biofuel policy has become more salient recently under the rubric of "rebound effect" discussed below (Smeets et al. 2014) .

2.3 *Total GHG Discharge as a Policy Criterion*

Even more tacit in this early discussion was the assumption that a unit reduction in GHG discharge, no matter how or when achieved, was as good for the climate–or for some broader measure of policy objectives, like "social welfare" broadly conceived–as reducing any other unit. This would mean that total GHG discharges up to an analytic time horizon, standardized to g CO_2e MJ^{-1}', would measure fuel systems, climate effects well enough to direct policy.

3 Land Use Change and Global Warming Intensity

In 2008, important papers (Fargione et al. 2008; Searchinger et al. 2008) extended the LCA concept for biofuels outside the production sequence framework, recognizing the central phenomena that (i) incompatible "uses", including natural vegetation, compete for a fixed land resource, (ii) fairly inelastic, international, food commodity markets transmit causal price signals from agricultural interventions around the world with behavioral consequences, and (iii) bringing land into cultivation from wild or pasture conditions releases significant amounts of GHG when existing plant materials–especially including very high carbon stock forests–burn or decay. The authors of these papers claimed that if new biofuel is driven into the market by any mechanism, a GHG discharge is triggered that is not recognized by ALCA, but that should be attributed to that biofuel just like the tractor's diesel consumption.

Conventionally, this discharge is characterized either as *direct land use change (Fargione)*, as non-cropland is planted with a biofuel crop, or *indirect land use* *change (Searchinger)* to describe the changes that are induced anywhere away from the biofuel production site. ILUC is commonly triggered by a sequence of "falling dominos" as different crops succeed each other and eventually forest or pasture is cultivated for (usually) a food crop. Generally the size of this discharge is estimated by an economic model that transmits a biofuel production increase shock into markets affecting natural and other land cover, and a land use model relating changes in cover to GHG discharges.

This chapter is not mainly concerned with the science of ILUC estimation, but the logic can be summarized in the following greatly simplified vignette (see, for example, Hertel et al. 2010 for a full explanation). In order to obtain feedstock for new corn ethanol production, a US refiner must outbid parties who wish to use corn for feed or food. This increases the price of corn, and some farmers grow less soybeans and more corn, while corn and soybean exports decrease. Remaining world demand for corn and soybeans induces conversion of pasture to crop production in (e.g.) Brazil, and conversion of Amazon forest to cattle raising to replace the lost pasture. At the same time, overall food consumption decreases. The land required for the new corn ethanol production is thus supplied by a combination of increased yields on existing cropland, reduced food consumption, and land conversion from natural conditions (Searchinger et al. 2008).

One result important for the present discussion is the broad agreement that a climate effect "of making and using more of a fuel" not visible in an ALCA should "count" in assessing policies promoting particular fuels. Standard analysis of biofuels' climate effects has thus been expanded from ALCA to a *consequential* life cycle analysis (CLCA) whose analytic boundaries comprise predictable effects of a change in practice, whether inside or outside the production process. CLCA importantly changes the substrate of which GHG discharge is properly predicated, from a *substance* such as bioethanol, to an *action*, such as making and using some amount more of the substance, or a *policy*, such as subsidizing use of that substance, and this change fits awkwardly at best with the basic structure of a lot of biofuel policy, both in implementation and advocacy (Plevin et al. 2013).

Furthermore, ILUC has turned out to be not just another CLCA-scored dimension, but because of distinctive qualities of its occurrence and analysis, it has drawn both scholarly and (to a lesser degree) policy attention to climate issues requiring new kinds of analysis and policy tools. Specifically,

- The GHG discharge, and consequently also the climate-forcing, time trajectories from using a biofuel and either an energy-equivalent *or a GHG-discharge-equivalent* amount of fossil fuel are very different (Kendall et al. 2009; O'Hare et al. 2009; Anderson-Teixeira and Delucia 2010; Levasseur et al. 2010; Cherubini et al. 2011).
- ILUC cannot be observed, but only predicted by complex models with many parameters, so its estimates are subject to wide uncertainty bands (both ways) (Plevin et al. 2010; Laborde and Valin 2012 Plevin et al. 2015).
- ILUC discharges are not unique to biofuels, but result from any activity that competes with food (at least commodity food) for land.

4 Application of GHG Estimation

Before turning to the lessons of the ILUC debate as presented by the foregoing qualities, it is worth reviewing who can or should do what with estimates of direct and indirect GHG discharges. There are at least four salient decisions facing players in this biofuel theater. For policymakers:

- Should use of a given type of fuel be encouraged in place of (typically) fossil fuel, especially for transportation where liquid fuels have especially high form value? If "yes",
- What policy mechanisms (subsidy, performance obligations, quantity obligations, etc.) should be used as levers, and how vigorously?
- What properties of which fuels qualify them for such leverage?
- And for participants in the market: what amounts, of what kinds of fuel, should I buy and use at current prices and under current rules?

Presumably, policymakers should make those decisions to maximize social welfare, and market participants will seek to maximize profit or utility. As the decisions at hand differ by locus and possibly purpose, we should note that different actors might rationally require different measures, even under the rubric of "climate effect", for different kinds of choice. The concept of "life cycle GHG emissions", for example, can be operationalized many different ways, producing different ratings and fuel preference orders, as is the case for regulatory rating systems implemented in California, British Columbia, and Europe (Plevin et al. 2013).

This decision structure highlights the comparative nature of the choices under consideration. Even if a GWI could be assigned to a given fuel with complete and uncontroversial certainty, no action by anyone follows from such a number; what matters is at the least, a pair of such measures for two competing fuels. For practical purposes, a judgment that "fuel A is good for the climate" is vacuous or depends on a tacit counterfactual: since the production of biofuels results in positive GHG emissions, these fuels contribute to GHG emission reduction only to the extent that their use results in the avoided use of fuel systems with greater GHG emissions (Plevin et al. 2013). In the pages that follow, we will for convenience talk about criterion measures that might be predicated of a specific fuel, or policy, but we assume that such measures are always to be used to compare alternatives.

More important is a growing recognition that the climate effects of expanding production and use of a given fuel are not properties of the fuel, or even the industrial system that produces it (DeCicco 2012), but of the policy that induces the change in fuel use. Important climate effects follow from factors that cannot be captured in a description of a fuel's life cycle (Plevin et al. 2013).

5 Uncertainty and Choice

As we observed above, one of the salient properties of ILUC measures is their inherent uncertainty (Plevin et al. 2015; Khanna and Crago 2012). Different modelers have found different values for what is putatively the same thing (Warner et al. 2013), and increasingly, analysts are providing not only central estimators of these values but also explicit estimates of the uncertainty associated with them (Plevin et al. 2010, 2015; USEPA 2010; Laborde and Valin 2012). Because ILUC cannot be observed directly, and because the scientific basis (parameter values and structure) of the models used to estimate it is always incomplete, this uncertainty has shown itself to be refractory: a decision-maker's best knowledge of the ILUC for a given fuel is and will always be a probability distribution whose variance is neither enormous nor small enough to ignore (Plevin et al. 2015).

What policymakers should do with this uncertainty was a topic of debate from the initial discovery of ILUC: one line of political advocacy, fortunately much less pursued than it was a few years ago, has been the remarkable idea that because ILUC is not known with complete precision or accuracy for any fuel, policymakers should act as though it is known with certainty to be zero (Plevin et al. 2010).

5.1 Decision Theory as a Framework

The formalism of statistical decision theory provides a good framework in which to think about the implications of uncertainty for a policy decision (Raiffa and Schlaifer 2008). Consider, for example, the relatively simple context of a *low carbon fuel standard* (LCFS) *a la* California. This policy requires the average GWI of California vehicle fuel to fall incrementally over a decade, and its mechanism of operation requires that the state's Air Resources Board (ARB) assign each fuel i in the California market a GWI G_i that is used to calculate an energy-weighted average GHG intensity for each fuel wholesaler's annual production (CARB 2009). The ARB has determined that this GWI shall be the sum of (i) a fuel's average production chain GWI as calculated using a version of GREET (Wang 1999), and (ii) the marginal ILUC emissions induced by increasing production of this fuel, as predicted using an adaptation of the GTAP model (Hertel 1997; Taheripour et al. 2008) combined with a GHG accounting model (Plevin et al. 2014). We note that both the production chain GWI and ILUC are uncertain, and that the sum of these two quantities estimated this way, though both measured in g CO_2e MJ^{-1}, may conceal some important double-counting. (Plevin 2015) Other jurisdictions confront the uncertainty in ILUC in the context of other consequential regulatory actions. For example, USEPA is obliged by statute to recognize ILUC, but only needs to classify fuels into a few categories based, in part, on GWI to implement the Renewable Fuel Standard (USEPA 2010), but the principles and issues presented by the seemingly straightforward LCFS apply in those cases as well. Our focus for the moment is on the uncertainty surrounding the "true value" of the ILUC emissions induced by the increased production of any fuel.

In the decision theoretic framework, a *decision-maker* must select one of several possible *actions* and will experience a payoff that depends on (i) the chosen action and (ii) the *state* of a system which is known to him only imperfectly, for example the value of a random variable that cannot be observed, if at all, until after he commits to an action. The decision-maker can assign a *probability density function* (PDF) to the possible values of the random variable. The simplest corollary to this exercise is a bet that pays off if the decision-maker correctly chooses the number or color of the next roulette wheel spin.

The key policy implementation step in the LCFS case is thus selection of operational values for the G_i's that fuel wholesalers must use in their reporting. If ILUC were a feature of a stationary system, and modeling presented an extremely narrow PDF for a fuel's ILUC emissions, like the distributions laboratory analysis gives us for things like a fuel's specific gravity, the implied action might seem to be to choose any of the very closely spaced central estimators of this value, such as the mean, as G_i, for each fuel. However, there is no unique, "true" value for ILUC emissions; there is a range of plausible "true" values obtainable under alternative futures (e.g., under different fuel price and policy regimes, and other stochastic developments in the economic system). A unitary decision-maker might assign subjective probabilities to these alternatives and combine them to produce a single PDF, but this PDF would only represent one decision-maker's belief about the "true" value for ILUC emissions. Nor is there consensus on how to combine ILUC emissions with the remainder of GHG emissions associated with producing and using a given fuel to produce a summary GWI. Under these circumstances, the choice of action becomes more complicated and entails attention to the decision-maker's belief about the respective *cost* of taking the right and wrong actions.

ARB's best information about the ILUC emissions associated with a fuel as defined by its chosen modeling framework is currently a PDF that has substantial variance and is also asymmetric, with a longer tail on the high side (Plevin et al. 2010; Plevin et al., 2015); including other estimates from other models as data in a Bayesian framework would increase this variance. Decision theory provides some standard results about the statistic of such a PDF that should be "bet" on, all of which depend on the *cost of being wrong* as a function of the error $e = G^*_i - G_i$. Usually the decision criterion is to minimize the expected loss, defined as appropriate for the decision. For example, if it is as bad for G_i to be ten grams per megajoule below G^*_i as ten grams above, and twice as costly to be wrong by twenty grams as ten (symmetric linear cost function), the median of the PDF of G^*_i is the optimal choice; if losses are quadratic in the absolute error, the best choice is the mean.

5.2 What Does It Mean to Act Conservatively?

The cost, however defined as the policy objective, conditional on $G*_i$, of choosing one or another value for G_i, is probably not so simply related to e. In the first place, it may not be symmetric. In the second place, what happens in the world when the regulator issues an operating value depends on the response of a large, complex system to that choice: when the regulator publishes a schedule of G_i's the fuel economy responds with a quantity of total fuel consumed and a mix of component fuels that depend on the prices of those components and the compliance benefit each provides. The correctly formulated decision in principle is to choose a vector of $\{G_i]$ that maximizes the objective

$$O = E\left[V\left(\{G_i\}, \{G_i^*\}, R\{G_i\}\right)\right],$$

where V is the net benefit of the outcome, and the expectation is not only over the probability distributions of the $G*_i$ but in principle also of R, the response of the economic system to the choice of the G_i's. We can relax the assumption that policy should maximize this expectation, and consider alternative rules such as minimizing maximum loss, but the complexity of the decision is not much reduced.

For the moment, let us assume that V is adequately measured in total GHG discharge in the California fuel system over thirty years (we relax this assumption below). If a given G_i is 10 g CO_2e MJ^{-1} below the real value, "too much" of fuel i will be used: are we indifferent between that outcome and using "too little" if G_i is too high, even if the extra GHG emitted is the same either way in the first year? It might appear so, but ILUC is more or less irreversible, and land use change has important biodiversity costs. Correcting the fuel mix when better information is available does not retrieve either of these costs, while underutilizing biofuel for a year only causes a year's worth of over discharge. Perhaps choosing G_i for biofuels on the "high side" is the prudent path.

On the other hand, giving current biofuels too high a G_i may cripple the development of an ethanol infrastructure that would greatly enhance the climate benefits of future cellulosic ethanol, and the safe action is one that deliberately advantages biofuel. Either way, the problem for the LCFS regulator has thus evolved far from turning the crank on the best available model and implementing "the number" it delivers, and the lever that forced it in this direction is the refractory uncertainty in the magnitude of ILUC emissions.

Lest the foregoing discussion seem arcane, we emphasize that choosing a best estimate (whatever *best* means) of the "true value" of the global warming intensity (e.g., g CO_2e MJ^{-1}) of a particular fuel is not the same as implementing a value for that fuel's GWI to be used in a regulation. Allowing an asymmetric cost function to displace an *operational value* from a *most likely* value is completely conventional in health and safety contexts. For example, a structural engineer is legally obliged to use a value, called a *design strength*, in sizing a steel beam that is much smaller than the average failure strength of such beams in a test laboratory.[2]

Policy debate about climate stabilization generally recognizes, though awkwardly, the need to compare the costs of being wrong in each direction; that is the implicit basis of a debate about whether to protect the current economy by waiting for more evidence of climate change, or move quickly to avoid possible catastrophes. The analysis of biofuel policy implementation, however, is only beginning to accommodate the implications of significant uncertainty in the size of ILUC emissions (and other GHG releases) and the need to consider what the "safe side" of regulatory practice is.

5.3 Decision-Making in the Real World

Our decision model departs significantly from the real world of the regulator. Decision theory assumes a unitary decision-maker who can assess his probabilities for each state of the world, and who knows his utility function over outcomes. In fact, energy policy is made by multiple actors in a political environment, actors whose utility structures differ and who have different data and therefore different probabilities for states. Sharing data helps bring these probabilities to be more alike, but to make people's outcome/payoff functions match requires deal-making and mechanisms to share gains and allocate risk.

If the uncertainty in the climate effect of a fuel (or policy) were owing only to variation in estimating predictive models' parameters, or even uncertainty about the environment in which the models' future would unfold, it would be relatively tractable. However, there is no consensus model of the global economy. The existing modeling systems vastly simplify a global economy, and rely on many assumptions known to be false in the real world (e.g., perfect competition, perfect information flow, market clearing, no savings, no unemployment, entire aggregated market sectors respond in unison to price changes). Key assumptions differ across models primarily because none has been shown to be superior to others (Ackerman and Nadal 2004; Laborde and Valin 2011). Consensus on how to model global climate impacts is not forthcoming, and events, in a world without control conditions, will not make it possible to observe which model is "best" even after years of

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[2] Safety factors are common throughout engineering, not only forcing us to behave as though materials are not as strong (and chemicals not as safe) as we know they are, but to overestimate likely loads (probably the most uncertain dimension of a structure's operating environment). We do this because we recognize a cost function in which a beam that is too weak will kill and injure people, or at least cause very expensive damage to property, while a beam that is too strong will only make a building cost more. Where cost functions have a different degree of asymmetry, for example where the structure in question is an airplane rather than a building and extra weight is especially costly, we knowingly adopt smaller safety factors; for brittle materials whose failure is not preceded by deformation that would warn users to evacuate, we adopt larger ones.

observation. Consequently, epistemic uncertainty about how "correct" any given model really is dominates the usual sources. In this situation, policymakers are well advised not to expect that more science will relieve them of the hard work of making choices that are deeply political and judgmental.

These conditions are not a counsel of despair, however. In the first place, a policy does not require everyone to agree on everything, but only on the policy to be adopted. In the second place, decisions properly framed are inevitable: it is not possible not to have any fuel/climate policy, or to have more than one at a time, so society has to bite the bullet and make choices. Third, public deliberation moves more fluidly if stakeholders' concerns can be recognized and correctly described. Also, while model predictions of ILUC vary for any given fuel, the finding that ILUC significantly reduces the LCA-estimated advantage of biofuels is consistent across a wide variety of approaches (Warner et al. 2013). Putting the issue of prudence and safety factors forward in policy implementation, as the uncertainty issue asks us to do, is likely to make it easier to analyze policy choices than forcing them to be debated as though we know more about $G*_i$'s than we really can.

6 Discharge Trajectories and Time

To this point, we have treated the costs and benefits of using one fuel or another as being adequately measured by the sum of life cycle GHG emissions. However, a key difference between ILUC and direct emissions' respective time trajectories of release, forces us to look behind that assumption. We find that total GHG is neither an unambiguous benefit measure nor serves as good of a proxy as it might appear.

One important time consideration that we will not examine closely here is presented by the different atmospheric lifetimes of GHGs with different radiative efficiencies. Under the Kyoto Protocol, different greenhouse gases are combined into a single "CO_2-equivalent" value using *global warming potentials* (GWPs) which represents the radiative forcing produced by 1 kg of a gas relative to that produced by 1 kg CO_2 over a chosen time horizon. For the present discussion, GWP scaling will suffice, but it is notable that the GWP for a gas other than CO_2 changes significantly depending on the time horizon used to construct the GWP (Ciais et al. 2013) and that counting a discharge with a GWP whose period extends beyond an analytic horizon (for example, using GWP_{100} for a discharge 50 years before the analytic horizon) is intrinsically incoherent. More important for the present discussion, what summing GWPs over a chosen analytic horizon does not account for is the *forcing at a given time as a function of when the GHG is emitted* during that analytic period. An alternative CO_2-equivalency metric, Global Temperature Potential (GTP) estimates the relative change in temperature produced *at a future point in time* (note that the point selected is a policy decision) by non-CO_2 gases or particles relative to the temperature change caused by an equal mass of CO_2 (Shine 2009; Tol et al. 2012; Ciais et al. 2013).

To implement a program like the LCFS, or to determine which of two fuels is "better" (perhaps to subsidize or regulate substitution of one for another) imposes the non-negotiable discipline of a scalar measure of merit. Multidimensional metrics are useful for many purposes, but a vector can only be *larger than* another vector if a defined function maps both into real numbers. From the start, ILUC analyses confronted a dimensional accounting problem posed by time of discharge: most of the emissions from ILUC from an increase in biofuel production occurs at the onset of the new production, is only partially and very slowly reversible, and is proportional to the size of the increase no matter how long it is sustained, while the other GHG releases of the biofuel system, and of the fuels for it is usually assumed to be substituted, are almost all proportional to, and occur with, the ongoing fuel use. The accounting problem of combining the one-time ILUC discharge with the ongoing direct discharges is analogous to the problem of combining the one-time capital cost of a factory with ongoing labor and materials costs to calculate the "cost of an item" produced in it. As they have different units (g CO_2e MJ^{-1} of additional *production capacity* for the first and g CO_2e MJ^{-1} of *fuel use* for the second) ILUC and direct discharges cannot be summed directly.

The early convention for including ILUC emissions in fuel GWI assumed a production period, implying a total quantity of biofuel produced over that production period, and allocated the ILUC discharge linearly over this quantity: on the assumption that the ethanol production triggering ILUC would continue for 30 years at constant fuel yield, for example, Searchinger et al. added 1/30th of the ILUC release to direct corn ethanol discharge, a calculation the California ARB still uses (Searchinger et al. 2008). In contrast, EU regulations "amortize" ILUC emissions over 20 years (European Parliament 2009). Obviously the assumed length of the production period has a large effect on the unit ILUC discharge value assigned to a fuel this way, and should: if an ethanol production increase lasted only a year or two, its GWI counting ILUC would be enormous. So far no persuasive analysis has demonstrated the "correct" production period to use in this calculation: should it be the average lifetime of an ethanol biorefinery? The time until widespread electrification of the fleet, or hydrogen fuel cells, or natural gas, make liquid fuels obsolete? More fundamentally, the choice of time horizon is not a scientific question but a political decision informed by a variety of kinds of market expertise. Whatever value is used, facilities may in fact produce more or less fuel. There's simply no way to know this in advance.

Another early analysis of land use change dealt with this problem as though the initial land use change discharge burdened the biofuel with a "carbon debt" that it repaid over time through its GHG advantage over fossil fuel in direct emissions (Fargione et al. 2008). But this number gives a regulator little help in implementing a policy like the LCFS. A fuel with a shorter payback period is "greener" than one with a longer period, *ceteris paribus*, but the payback period does not translate easily into a GWI score. In any case, what is an "acceptable" payback period, and what if the ethanol production in fact does not continue that long? And how would we compare biofuels to fossil fuels, which have an infinite payback period?

Melillo et al. (2009) suggested that if a carbon charge can be instituted as an overarching climate policy, biofuels should simply be taxed on their total first-year discharge GWI, including ILUC, leaving it to the financial system and subsequent year's market advantage to pay a literal carbon debt.

6.1 Discharge Trajectories

The implications of differing discharge profiles for different fuels are even more complicated than capital cost accounting, because GHGs have different atmospheric lifetimes and cause warming while they remain in the air. A gram of CO_2 emitted now will have caused more accumulated forcing (temperature increase) by 2050 than the same gram emitted in 2040 (Kendall et al. 2009; O'Hare et al. 2009). Figure 1 shows discharges, atmospheric GHG concentration (cumulative discharges less decay), and cumulative forcing over time for a fossil fuel and an energy-equivalent quantity of biofuel, *each of which emits the same total amount of*

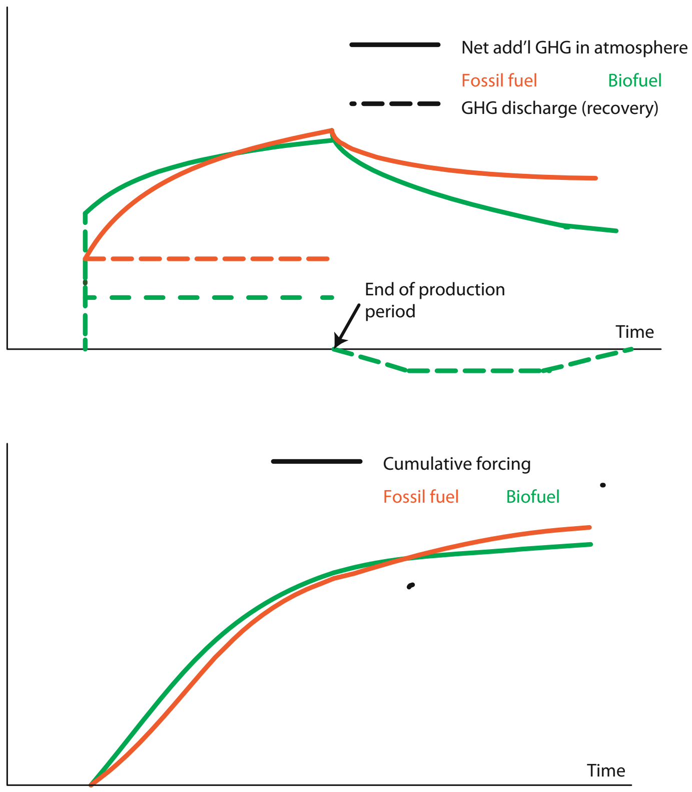

Fig. 1 Time discharge profiles and forcing for a biofuel and a fossil fuel

CO_2 during the production period. For example, this might represent a biofuel with 60 g CO_2e MJ^{-1} (of fuel use) direct discharge and 1050 g CO_2e MJ^{-1} (of production capacity) for ILUC, produced over 30 years, and gasoline with 95 g CO_2e MJ^{-1} direct discharge used for the same period. In this sketch, biofuel production begins with ILUC, fossil and biofuel use continue for the production period, and the land where ILUC occurred then begins to regrow to natural conditions, recapturing CO_2 from the atmosphere (if it does not, the failure to recapture cannot be "blamed" on the biofuel). Meanwhile, emitted GHGs accumulate in the air and decay out. This picture of discharges solves the problem of combining ILUC with direct discharges, but only by combining them into one or another function of time, not a scalar measure.

Note that with our assumption of regrowth sequestration, MJ-MJ substitution of any biofuel whose direct GHG discharges are lower than a fossil fuel's, grown for any production period, will eventually have emitted less total GHG than the fossil fuel, and eventually cause less cumulative forcing. However, "eventually" may be a very long time. The significance of this figure is that all the fossil and biofuel pairs of time trajectories cross at some time in the future.

Which fuel is more climate-friendly? If the trajectories did not cross, the answer would be simple (though how much "greener" the biofuel is would still require analysis). But as they do, the answer to that question requires a policy decision about what criterion–that is, *what scalar function of which trajectory*–should be used to assess them. Even if the criterion is as simple as "total GHG emissions", it remains to choose a time up to which that integral (of emissions) is evaluated, because moving the *analytic horizon* beyond the crossing point will switch our preference between the fuels.

Neither policy analysis nor political theory gives us a solid basis on which to choose such a horizon. Perhaps the right date is the end of the current governor's or president's term? A date by which nearly everyone now alive on earth will be dead, say 100 years? The most likely date by which the Greenland ice cap will have melted into the oceans? The 500 years of the IPCC's GWP_{500}?

Using very long analytic horizons without discounting (see below) leads to absurd policy implications. A 500-year analytic horizon, for example, would mean that Drake should have considered the effects, including climate effects, on a twenty first century industrial world he could not even imagine, with the same weight as its contemporaneous effects when he burned the Spanish timber stores at Cádiz (Mattingly 1989).

Even if some operational time horizon were adopted within which discharges were simply summed, we would still not be out of the woods with a defensible, coherent practice. Recall, in particular, that the reason we care about GHG emissions is not for their own sake but for their climate-forcing effect: if fuels policy is about global warming, it seems important if a policy minimizing GHG emissions for a given period actually results in more warming for decades, and it would seem appropriate to infer the criterion from the forcing, or at least the cumulative atmospheric concentration, trajectory rather than discharges alone. No current policy implementation, to our knowledge, does this. In the context of crop-based biofuels with ILUC, it is important that moving the criterion assessment from total discharge, to cumulative atmospheric GHG, to cumulative forcing successively extends the time until a given biofuel will score "better" than a fossil fuel.

Since the original paper on discharge time accounting for ILUC (O'Hare et al. 2009), several improvements and alternative approaches have been presented, both for liquid fuels (Kendall et al. 2009; Levasseur et al. 2010) and forestry, where long intervals between harvest/combustion (discharge) and regrowth are important (Cherubini et al. 2011). However, policy design and implementation in this area still use simple discharge summation over an arbitrary production period as the operational criterion.

6.2 Discounting

There remains the issue of comparing costs incurred now with costs in the future. If we score fuels on the basis of cumulative forcing–certainly a better proxy for social cost than emissions totals–over a long period, we have to ask whether we are really indifferent between increasing planet temperature by a degree F during the decade 2050–2059 and doing the same for the decade 2020–2029; both would add the same amount to the cumulative index as of $t > 2060$. Economics provides us a model, exponential discounting, with which individuals and societies conventionally compare costs and benefits now to costs and benefits in the future. This model, widely used in benefit/cost analyses of all kinds, counts a consequence with an economic interpretation (not necessarily a money value) in year t with $(1 - r)$ of the weight of the same consequence occurring in year $t - 1$. Here r is a discount rate, whose size is itself a policy decision requiring analysis. With this accounting, things that will happen later count less in a benefit/cost assessment than things that happen sooner.

A nonzero discount rate can portray a variety of considerations, including so-called "pure time preference," the psychological principle that we prefer benefits sooner than later; actual financial trading outcomes in markets across time; and uncertainty about predicted events. (A further complication is introduced by the *social cost of carbon* (SCC) correction advocated by a US government working group (Interagency Working Group on Social Cost of Carbon 2013), which counts GHG discharges as *more* costly (before discounting) insofar as they occur later, because they affect a larger population and economy. Analytically, this correction can be captured by subtracting the annual SCC growth rate from the chosen discount rate.)

To date, almost no researchers have incorporated discounting in their models of discharge time profiles. We consider this an important opportunity to improve both scholarship and policymaking. Even though the right discount rate for such analyses is arguable, using a rate anywhere in the plausible range of about 3–10% greatly increases the plausibility and legitimacy of this kind of model.

Note that discounting only applies to economic values (not necessarily values measured in money, though they have to be measured in some consistent way) that can actually be traded across time. The classic example is a financial instrument such as a bank deposit that trades control of funds from one time to another. It is not generally meaningful to discount physical quantities: a bucket of water in January is worth less, not more, than the same bucket in June if its purpose is to water a summer garden; if used to put out a fire in January, discounting does not measure how much more valuable it is then than in June. Consequently, discounting GHG discharges across time implicitly assumes that those quantities are close enough proxies for social cost incurred at the time of discharge. For effects spread over decades and centuries, this is not the case and discounting should be applied to more appropriate indicators like annual forcing.

Even a small discount rate accumulates large effects when time periods get long; for example, discounted at 3%, a dollar's worth of benefits in 2034 counts as only about 50c now; a 2114 dollar, 5c. At 5%, a 2064 dollar is worth less than 8c now. Instead of the kind of "angels on the head of a pin" reasoning required to contemplate 100 or 500-year analytic horizons, it seems wiser to accept infinite horizons and discount the relevant trajectories. This automatically, elegantly, and defensibly makes events in the conjectural, far, future count for very little and makes consequences during futures we can reasonably analyze salient. For an example of the effect of discounting on discharge profiles, see O'Hare et al. (2009).

The "discount rate debate" in the climate context has mostly been focused on the trade-off between incurring costs for stabilization now and receiving benefits years or decades in the future when climate change would otherwise be more severe. Advocates of faster investment in stabilization typically argue for smaller discount rates. It is not possible to provide a complete analysis of the role of discounting in climate policy here (Schelling 1995; Guo et al. 2006; Stern 2006; Nordhaus 2007); for the present context, we accept that conventional discounting is required for rational policy analysis.

6.3 *Time Displacement Accounting*

We return briefly to the question of the assumed production period for a biofuel system, which as noted above, is extremely difficult to estimate with confidence and greatly affects the unit contribution of ILUC to a biofuel's GWI. An insight from Kløverpris and Mueller (2012) provides an opportunity to sidestep this issue completely. They note that agricultural expansion for all purposes (such as increased food demand) is greater in a given year than the expansion needed for projected biofuel production, so the effect of producing biofuel for one season is merely to cause the biofuel ILUC to occur a year earlier than it otherwise would.

The model they construct on this insight has many of the problems discussed above, but treating ILUC as merely accelerating land clearing that is already going to happen leads to a shortcut calculation we can call *time displacement accounting* that avoids almost all the complexity of complete time profile models and does not require an assumed production period. Note that in this framing, all the discharges from the biofuel and comparable fossil fuel occur within a year, so most of the issues raised by the difference between forcing and discharge trajectories are avoided: for discharges less than a year apart, any pound of GHG (scored as IPCC GWP), however discharged, has about the same social cost as any other, so we can reasonably use total GHG discharge in GWP terms over this short period as proportional to social cost.

Causing the ILUC to occur a year earlier than it otherwise would has a social cost of about r times the social cost of the ILUC, just as paying a dollar a year early costs about 5c at 5%. Furthermore, this effect is independent of production period; no matter how long or briefly production continues (again, assuming land conversion worldwide annually exceeds biofuel land needs), each year of production independently causes this acceleration of clearing discharge. Accordingly, it is a reasonable approximation and a tractable protocol to simply multiply total ILUC by r and add it to direct emissions to calculate GWI: at 5%, for example, a fuel with ILUC of 500 g and direct emissions of 60 g could be reasonably assigned a GWI of $60 + (0.05 \times 500) = 85$ g. The uncertainty in estimating ILUC itself remains to be dealt with as in the previous section, and a regulator needs to settle on an appropriate discount rate (which we have argued is the case no matter what) but much of the difficulty in accounting for time issues is avoided.

6.4 Discharge Profiles and ILUC Beyond Biofuels

We now briefly note two implications of the analysis of ILUC whose application is broader than biofuel GWI estimation. The first is that, as Bruce Dale has noted in conversation, ILUC is not just caused by biofuel production but by anything that competes with food for land. This includes state parks, highways on arable land, and, importantly, suburban development on what would otherwise be farmed. Developers like to build on flat land, and every acre of (say) Chicago suburbs displaces corn from food markets and must have an even larger ILUC discharge (because it produces no feed byproducts) than an acre of corn used for ethanol instead of feed or food. The ILUC effect of putting farmland to any other uses deserves analysis and recognition in land use regulation.

The second lesson is that discharging GHG early in an energy production system causes more forcing for a long time than discharging it later. Nuclear power and hydropower, for example, are generally scored as low carbon, but the carbon discharges attributable to the large amounts of steel and concrete used to build dams and reactors occur at the very beginning of the systems' operational life and represent a much larger fraction of those technologies' total forcing than time-ignoring calculation of discharge quantities indicate.

7 Beyond ILUC

7.1 Non-land-Use Climate Effects

The big lesson from ILUC research–that everything in the world is connected, albeit with bungee cords rather than chains–presents policymakers and analysts with the likelihood that a large biofuel production increase will indirectly generate still other, non-land use, GHG discharges not counted in an ALCA. Two of these have been identified and shown to be considerable, the "rebound" effect on petroleum use and N_2O releases from increased fertilization of all crops. A strong case can be made that, if ILUC is real and should "count" in scoring fuels for climate, these other indirect climate effects should be counted as well (Khanna et al. 2011).

The rebound effect, sometimes called *indirect petroleum use change* (IPUC), is the failure of biofuel forced into a particular market, such as the US, to displace fossil fuel MJ-for-MJ worldwide (Rajagopal and Plevin 2013). The displacement that does occur reduces fossil fuel prices worldwide and increases petroleum use everywhere outside the biofuel program jurisdiction, an increase that partly offsets the climate gains from the domestic substitution. Estimates of this effect show it to be in the range of 30–70% of MJ-MJ substitution amounts (Hochman et al. 2011; Chen and Khanna 2012; Bento and Klotz 2014; Chen et al. 2014).

Increases in world food prices motivate farmers to increase yields, though the size of this effect, which is an important factor in the size of ILUC, has been strongly debated. One way to increase yields on existing cropland is to fertilize more heavily, especially with nitrogen, and this additional application generates added releases of N_2O, a potent greenhouse gas (Crutzen et al. 2007; Melillo et al. 2009; Reay et al. 2012).

7.2 Non-climate Costs and Benefits

Our discussion to this point has emphasized the difference between total GHG discharge and climate forcing, generally indicating that scoring biofuels by the latter (better) proxy for social cost makes them look less attractive from a climate stabilization perspective. But *social cost*, what policy should tautologically minimize, comprises much more than climate change, and biofuels expansion has important non-climate effects (as does fossil fuel extraction). Some of these dimensions of social cost are accounted for by the market prices of biofuels and their competitors and do not present an especially difficult problem. LCA, as we noted, counts the diesel fuel the corn farmer's tractor uses, but not the gas that cooks his eggs in the morning nor the cost of the eggs themselves (but see (Giampetro 2009) for analysis that incorporates energy costs of human labor). However, those costs, energy and other, are reflected well enough in the price he requires for his corn that we can consider them as accounted for in the social decision system.

Other costs, however, are not well reflected in market prices. One example is the effect on nutrition that the ILUC-estimating models present along with their land use results: using land for energy instead of food makes food more expensive and reduces its consumption (Searchinger et al. 2015) . Some of that reduction may benefit the overweight, or represent less meat in the diet of the middle class, and may be no great concern, but some of it is less food overall consumed by people who are already nutrition insecure. Biofuel production requires water that is scarce in some regions and usually not priced efficiently (Fingerman et al. 2010).

ILUC forest clearing not only releases GHG but also reduces biodiversity (Fargione et al. 2010). Changes in agricultural practices, especially in developing countries, affect social structures, rural employment, and local politics.

In principle, these additional costs (and benefits) of biofuel expansion and production should be included in analysis of the optimal choice of G_i discussed under uncertainty, and the GWI values chosen to minimize a discounted indefinite stream of net benefits of all (identifiable) kinds. Also in principle, or in theory, all of them could be estimated by modeling and otherwise, priced by the benefit–cost analysis heuristics and devices used to account for nonmarket goods, converted to units of GHG by a carbon price, and added directly into the GWI calculation for a fuel, thus measuring a variety of costs in units of GHG emissions in order to conform to the administrative structure of a fuel policy.

It is not clear that this would be a good idea, however, even if it were analytically tractable. The LCFS, for example, is subject to general expectations of increasing social welfare, but it is also a fuel carbon intensity policy. A constant challenge to implementing enacted programs with a specific scope and purpose is the natural desire of advocates of other goals to hang their own concerns on it, until the policy in question becomes an unmanageable, uncoordinated Christmas tree of good intentions. Nevertheless, and recognizing the risk of second-best problems, it does seem reasonable to integrate large non-climate effects into GWI measures where it can be done easily.

Two examples of this integration apply to food and water. Hertel et al. (2010) included in their analysis a model framework that constrained food consumption to remain constant during a biofuel production increase shock, and observed about 40% more ILUC. Using this value is not an improvement in the estimate of ILUC, but a policy decision not to "count" climate benefits resulting from food deprivation. A recent extension of the GTAP model for ILUC incorporated a restriction of agricultural expansion to regions where irrigation water is available, and ILUC increased about 27.5% (Taheripour et al. 2013). This amendment represents a better description of what using more of a biofuel will actually do in climate terms, not a judgment about the value of water.

8 Conclusions

Fuel policies affecting or depending on crop-based biofuels cannot properly be implemented or analyzed merely by summing GHG discharges as the designers of (for example) the LCFS or RFS implicitly assumed. The distinctive properties of the ILUC discharges induced by increased production of these fuels, especially including the refractory variance in estimates of it, and the mismatch between the GHG release profile of fuels with ILUC and competing fossil fuels, require attention to considerations that are still novel in both research and policy implementation. Among these are the cost of being wrong about a fuel's GWI and the failure of summed GHG emissions to reflect social cost when time trajectories of fuel GHG releases do not match.

It is too early to make firm recommendations for policy that account for all these implications of ILUC research. It may become necessary to redesign the policies substantially so they do not require fuel-specific GWI values. However, for the near future, and as research and policy design proceed, some action recommendations can be advanced with confidence.

First, the operational merit index for fuels—their GWI—should be more sophisticated about time than merely summing discharges over an arbitrary production period, preferably time-aware, and should be based on discounted cumulative forcing, with or without a social cost of carbon (SCC) correction. Implementing agencies should "bite the bullet" and (i) adopt a discount rate to address the time profile of ILUC emissions and (ii) analyze probable production periods to identify a sound assumption so as to make this possible. If forcing is discounted, it is not necessary to choose an analytic horizon. Alternatively, if the assumptions behind time displacement accounting are supported with further research, it is an acceptable and reasonably accurate heuristic that avoids much of the complicated time-aware modeling and can adequately compare fuels with GHG discharge estimates directly.

Second, implementing agencies should confront the issue of safety factor and prudence by (i) a systematic analysis of the cost of positive and negative errors in assigning GWI values, and (ii) an affirmative decision to choose values relatively far from the chosen GWI distribution's central estimators if that is indicated. Sophisticated approaches to addressing uncertainty and risk are therefore required to incorporate ILUC emissions into regulations properly (Plevin et al. 2013; Witcover et al. 2013).

Third, water and food considerations should be incorporated into GWI values (for policies where GWI is the operating mechanism) by means such as the constrained models described above.

Outside fuel policy, the lessons of ILUC for food-competitive land use and early discharge trajectories should be adopted into policy and climate analysis of energy sources.

Fourth, and generally: the most important lesson of ILUC analysis is that indirect effects of many kinds affect the climate benefits of fuel substitution, and effects such as rebound and noncombustion releases like methane from changing cattle and rice production, and N_2O releases from increased fertilizer use should be recognized and counted as well as possible.

Appendix: The LCFS Mechanism

The most uncertainty-intolerant biofuels policy framework at present is the California Low Carbon Fuel Standard (LCFS). The machinery of the LCFS is simple in principle: every fuel distributor in the state annually calculates his annual *average fuel carbon intensity* (AFCI) as a sum of the fuels combined into motor fuel, weighted by their respective officially published GWI values. If this AFCI is below the gradually declining standard specified in the regulation, the fuel distributor has allowances to sell to other distributors; if higher, he must pay a fine or purchase allowances.

Specifically, for a fuel distributor (approximately, wholesaler) j in year t who blends Q_i units of fuels $i = 1, 2, 3\ldots$ with GWI values G_i, the fine (or sale of credits) C_{jt} at a "price" of P when the standard is S_t will be:

$$\begin{aligned}\mathrm{AFCI}_{jt} &= G_pQ_p + G_bQ_b \\ C_{jt} &= \left(S_t - \mathrm{AFCI}_{ji}\right)PQ_i.\end{aligned}$$

In this example, p is petroleum and b is a biofuel, and calculations are all per MJ of available fuel energy (diesel and electric energy receive an adjustment to account for their higher efficiency). Assuming biofuel costs more than fossil fuel and it is not being blended to the limit to minimize product cost, a lower value for G_b will lead to cheaper compliance for the distributors. Though it was widely expected when the LCFS was developed that the main compliance paths for gasoline and diesel would be admixtures of maize ethanol and soybean biodiesel, the requirements of this policy are quite stringent. For example, if the "blend wall" for ethanol into gasoline is 20% (twice the current legal limit) and there is little market penetration of so-called E85 or flex-fuel vehicles (that can operate on any ethanol percentages) during the LCFS' period of operation, compliance with a 10% carbon intensity reduction from gasoline's 96 g CO_2e MJ^{-1} requires "45-g" ethanol, a value almost no current domestic bioethanol can be shown to have *even ignoring the indirect discharges from ILUC (Liska).*

References

Ackerman, F., and A. Nadal (eds.). 2004. *The flawed foundations of general equilibrium: critical essays on economic theory*. London; New York, Routledge: Routledge frontiers of political economy.

Anderson-Teixeira, K.J., and E.H. Delucia. 2010. The greenhouse gas value of ecosystems. *Global Change Biology* 17 (1): 425–438.

Bento, A.M., and R. Klotz 2014. Climate Policy Decisions Require Policy-based Lifecycle Analysis. *Environmental Science & Technology*.

CARB (2009). Proposed Regulation to Implement the Low Carbon Fuel Standard, Volume I, Staff Report: Initial Statement of Reasons. Sacramento, CA, California Air Resources Board: 374.

Chen, X., H. Huang, et al. 2014. Alternative Transportation Fuel Standards: Economic Effects and Climate Benefits. *Journal of Environmental Economics and Management*.

Chen, X., and M. Khanna. 2012. The Market-Mediated Effects of Low Carbon Fuel Policies. *AgBioForum* 15 (1): 1–17.

Cherubini, F., G.P. Peters, et al. 2011. CO_2 emissions from biomass combustion for bioenergy: atmospheric decay and contribution to global warming. *GCB Bioenergy* 3 (5): 413–426.

Ciais, P., C. Sabine, et al. 2013. Chapter 6: Carbon and Other Biogeochemical Cycles. Climate Change 2013: The Physical Science Basis. In *Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, ed. T.F. Stocker, D. Qin, G.K. Plattneret al., 1535 Cambridge, United Kingdom and New York, NY, USA, Cambridge University Press.

Crutzen, P.J., A.R. Mosier, et al. 2007. N_2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. *Atmospherique Chemical Physics Discussions* 7 (4): 11191–11205.

DeCicco, J. 2012. Biofuels and carbon management. *Climatic Change* 111 (3–4): 627–640.

Drabik, D., and H. de Gorter. 2011. Biofuel policies and carbon leakage. *AgBioForum* 14 (3): 104–110.

Ekvall, T., and B.P. Weidema. 2004. System boundaries and input data in consequential life cycle inventory analysis. *International Journal of Life Cycle Assessment* 9 (3): 161–171.

European Parliament 2009. Fuel Quality Directive 2009/30/EC. European Parliament, Official Journal of the European Union.

Fargione, J., J. Hill, et al. 2008. Land Clearing and the Biofuel Carbon Debt. *Science* 319 (5867): 1235–1238.

Fargione, J., R. J. Plevin and J. Hill 2010. The Ecological Impact of Biofuels. *Annual Review of Ecology, Evolution, and Systematics* 41: 351–77.

Farrell, A.E., R.J. Plevin, et al. 2006. Ethanol Can Contribute to Energy and Environmental Goals. *Science* 311: 506–508.

Fingerman, K.R., M.S. Torn, et al. 2010. Accounting for the water impacts of ethanol production. *Environmental Research Letters* 5 (1): 014020.

Finnveden, G.R., M.Z. Hauschild, et al. 2009. Recent developments in Life Cycle Assessment. *Journal of Environmental Management* 91(1):1–21.

Giampetro, M., and Kozo Mayumi. 2009. *The Biofuel Delusion*. London: Earthscan.

Guo, J., C.J. Hepburn, et al. 2006. Discounting and the social cost of carbon: a closer look at uncertainty. *Environmental Science & Policy* 9 (3): 205–216.

Hertel, T.W. 1997. *Global trade analysis: modeling and applications*. Cambridge: Cambridge University Press.

Hertel, T.W., A. Golub, et al. 2010. Effects of US Maize Ethanol on Global Land Use and Greenhouse Gas Emissions: Estimating Market-Mediated Responses. *BioScience* 60 (3): 223–231.

Hochman, G., D. Rajagopal, et al. 2011. The Effect of Biofuels on the International Oil Market. *Applied Economic Perspectives and Policy* 33 (3): 402–427.

Interagency Working Group on Social Cost of Carbon (2013). Technical update of the Social Cost of Carbon for Regulatory Impact Analysis. Washington, DC, U.S. Government.

Kendall, A., B. Chang, et al. 2009. Accounting for Time-Dependent Effects in Biofuel Life Cycle Greenhouse Gas Emissions Calculations. *Environmental Science and Technology* 43 (18): 7142–7147.

Khanna, M., and C.L. Crago. 2012. Measuring Indirect Land Use Change with Biofuels: Implications for Policy. *Annual Review of Resource Economics* 4 (1): 161–184.

Khanna, M., C.L. Crago, et al. 2011. Can biofuels be a solution to climate change? The implications of land use change-related emissions for policy. *Interface Focus* 1 (2): 233–247.

Kløverpris, J. and S. Mueller. 2012. Baseline time accounting: Considering global land use dynamics when estimating the climate impact of indirect land use change caused by biofuels. *The International Journal of Life Cycle Assessment* 1–12.

Laborde, D. and H. Valin 2011. Modelling Land Use Changes in a Global CGE: Assessing the EU biofuel mandates with the MIRAGE-BioF model. In *14th GTAP Conference*, June, 16–18, 2011. Venice, International Food Policy Research Institute (IFPRI).

Laborde, D., and H. Valin. 2012. Modeling land-use changes in a global CGE: assessing the EU biofuel mandates with the MIRAGE-BioF model. *Climate Change Economics* 03 (03): 1250017.

Levasseur, A., P. Lesage, et al. 2010. Considering Time in LCA: Dynamic LCA and Its Application to Global Warming Impact Assessments. *Environmental Science & Technology*.

Mattingly, G. 1989. *The Armada. Norwalk, Conn.* Easton Press.

Melillo, J.M., J.M. Reilly, et al. 2009. Indirect Emissions from Biofuels: How Important? *Science* 326 (5958): 1397–1399.

Nordhaus, W. 2007. The Stern Review on the Economics of Climate Change. *Journal of Economic Literature* 45 (3): 686–702.

O'Hare, M., R.J. Plevin, et al. 2009. Proper accounting for time increases crop-based biofuels greenhouse gas deficit versus petroleum. *Environmental Research Letters* 4 (2): 024001.

Plevin, R.J. 2009. Modeling corn ethanol and climate: A critical comparison of the BESS and GREET models. *Journal of Industrial Ecology* 13 (4): 495–507.

Plevin, R.J. 2010. Life Cycle Regulation of Transportation Fuels: Uncertainty and its Policy Implications. Ph.D., University of California—Berkeley.

Plevin, R.J., J.F. Beckman, A. Golub, J. Witcover and M. O'Hare 2015. Carbon accounting and economic model uncertainty of emissions from biofuels-induced land use change. *Environmental Science & Technology* 49 (5): 2656−2664.

Plevin, R.J., M.A. Delucchi, et al. 2013. Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation Benefits Misleads Policy Makers. *Journal of Industrial Ecology* 18 (1): 73–83.

Plevin, R.J., H.K. Gibbs, et al. 2014. Agro-ecological Zone Emission Factor (AEZ-EF) Model (v47). *Global Trade Analysis Project (GTAP) Technical Paper No. 34. GTAP Technical Paper*. West Lafayette, Indiana, Center for Global Trade Analysis, Department of Agricultural Economics, Purdue University.

Plevin, R.J., M. O'Hare, et al. 2010. Greenhouse Gas Emissions from Biofuels: Indirect Land Use Change Are Uncertain but May Be Much Greater than Previously Estimated. *Environmental Science and Technology* 44 (21): 8015–8021.

Raiffa, H., and R. Schlaifer. 2008. *Introduction to Statistical Decision Theory*. Boston: MIT Press.

Rajagopal, D., and R.J. Plevin. 2013. Implications of market-mediated emissions and uncertainty for biofuel policies. *Energy Policy* 56: 75–82.

Reay, D.S., E.A. Davidson, et al. 2012. Global agriculture and nitrous oxide emissions. *Nature Climate Change* 2 (6): 410–416.

Schelling, T. 1995. Intergenerational Discounting. *Energy Policy* 23 (4–5): 395–401.

Searchinger, T., R. Heimlich, et al. 2008. Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land Use Change. *Science* 319 (5867): 1238–1240.

Searchinger, T., R. Edwards, D. Mulligan, R. Heimlich and R. Plevin 2015. Do biofuel policies seek to cut emissions by cutting food? *Science* 347 (6229): 1420−1422.

Shine, K. 2009. The global warming potential—the need for an interdisciplinary retrial. *Climatic Change* 96 (4): 467–472.

Smeets, E., A. Tabeau, S. van Berkum, J. Moorad, H. van Meijl and G. Woltjer 2014. The impact of the rebound effect of the use of first generation biofuels in the EU on greenhouse gas emissions: A critical review. *Renewable and Sustainable Energy Reviews* 38 (0): 393−403.

Stern, N. 2006. *Stern Review on the Economics of Climate Change*. HM Treasury, Government Economic Service.

Taheripour, F., D.K. Birur, et al. 2008. *Introducing Liquid Biofuels into the GTAP Data Base. GTAP Research Memorandum*. W. Lafayette, IN: Purdue University.

Taheripour, F., T.W. Hertel, et al. 2013. The role of irrigation in determining the global land use impacts of biofuels. *Energy, Sustainability and Society* 3 (1): 4.

Tol, R.S.J., T.K. Berntsen, et al. 2012. A unifying framework for metrics for aggregating the climate effect of different emissions. *Environmental Research Letters* 7 (4): 044006.

USEPA 2010. *Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis*. Washington, DC: US Environmental Protection Agency: 1120.

Wang, M.Q. 1999. *GREET 1.5—Transportation Fuel-Cycle Model, Volume 1: Methodology, Development, Use, and Results*. Center for Transportation Research, Energy Systems Division, Argonne National Laboratory.

Warner, E., Y. Zhang, et al. 2013. Challenges in the estimation of greenhouse gas emissions from biofuel-induced global land-use change. *Biofuels, Bioproducts and Biorefining*.

Witcover, J., S. Yeh, et al. 2013. Policy options to address global land use change from biofuels. *Energy Policy* 56: 63–74.