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**On environmental lifecycle assessment for
policy selection**

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On environmental lifecycle assessment for policy selection

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Abstract

Lifecycle Analysis (LCA) has become an important tool for guiding regional or national policy actions to address global environmental problems such as climate change. LCA-based indicators of greenhouse gas (GHG) intensity of different fuels are being used to design long-term policies supporting renewable and alternative energy technologies. However, some of these technologies risk proving counter-productive to policy goals. An example is the debate about the environmental benefits of biofuels. Using biofuels as an illustrative example, we identify the structural reasons for the differences between two strands of literature on environmental benefits of alternative energy, namely, lifecycle analysis and economic market-equilibrium analysis. We explain why a policy planner cannot assume the potential environmental gains as revealed by a comparison between two LCAs as given while selecting policies to capture those gains. In other words lifecycle indicators are endogenous variables in the policy-selection problem. A capacity to compute lifecycle indicators as a function of economic variables and policy parameters can help policy planners better compare the implications of different policy actions.

Keywords: energy, environment, lifecycle assessment, general equilibrium, public policy

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1 Introduction

Lifecycle analysis or assessment (LCA) has become an important tool for guiding regional or national actions to address global environmental problems such as climate change (EPA, 2009a; ARB, 2009). LCA-based indicators of greenhouse gas (GHG) intensity of different fuels are being used to justify long-term policies supporting renewable energy. However, some renewable energy policies risk becoming counter-productive to the environment. An example is the debate about the environmental benefits of biofuels. Whereas LCAs initially suggested that biofuels such as corn ethanol can reduce GHG emissions (de Carvalho, 1998; Wang, 1999; Sheehan et al., 2000; Farrell et al., 2006)¹, subsequent research which uses economic equilibrium techniques to simulate the future outcomes under current biofuel policies suggests otherwise. The latter literature predicts that these policies will accelerate GHG emissions for at least a decade or two before contributing to net emission reduction (Searchinger et al., 2008; Melillo et al., 2009; Dumortier et al., 2009; Havlik et al., 2010; Hertel et al., 2010). In this paper, we identify the structural reasons for the differences between these two strands of literature on the environmental benefits of renewable fuels, namely, lifecycle analysis and economic (partial and general) equilibrium analysis. Using biofuels as an illustrative example we explain why, a policy planner cannot assume the potential environmental gains as revealed by a comparison between two LCAs as given while selecting policies to capture those gains. In other words, the lifecycle footprint of any given technology is an endogenous variable in the policy selection and planning problem. We therefore argue that policy makers should compare technologies taking into account the differences in their trajectories over time as a result of innovations as well as changes in market conditions under different policy regimes. We identify the need for a new modeling framework, which computes the lifecycle environmental footprint of any given technology as a function of the economic variables and policy parameters.

Economic theory suggests that the cost-effective strategy to address a global environmental problem such as climate change is through a globally consistent GHG policy that treats emissions from all human activities alike (Stern et al., 2006). However with political agreement proving elusive even at a national-level, not to mention the goal of a global consensus on GHG emissions, regional initiatives such as Californias Global Warming Act (AB32) and Regional Greenhouse Gas Initiative (RGGI) in the US North-east, are assuming a major role in addressing climate change. To complement these regional policies and in several cases to substitute the lack of one, targeted policies such as Renewable Portfolio Standards (RPS) for electricity, Renewable

¹A notable exception to this consensus was Pimentel and Patzek (2005) who argued that all first generation biofuels are more carbon-intensive than gasoline or diesel, even when market mediated emissions are not taken into account.

fuel standards (RFS), and Californias State Bill 375 that targets emission reduction from land use are being implemented by provincial and local governments. Such regulations that target emission from only a subset of GHG generating activities can sometimes prove counter-productive. For instance, reducing tail-pipe GHG emissions from automobiles by substituting fossil fuels with biofuels increases emissions from agriculture (from the use of farm chemicals, farm machinery, land use etc.) and industrial processing i.e, conversion of crop to fuel. Similarly, for almost any other alternative to conventional fossil fuels including oilsand, liquids from coal and natural gas, hydrogen and electric propulsion technologies, a large amount of lifecycle emissions are not embodied in the fuel but occur upstream. Depending on the type of resource (say, light sweet crude or heavy crude), the conversion process (for instance, strip mining or *in situ* extraction of oilsand) and type of energy used in processing (for instance, whether coal or natural gas) lifecycle emissions are highly variable (MacLean and Lave, 2003; Brandt and Farrell, 2007; Rajagopal and Zilberman, 2008a; Charpentier et al., 2009; Spatari et al., 2010).

This provides a rationale for the use of Lifecycle Assessment (and related frameworks such as Input-output analysis and embodied energy) in public policy. LCA is a technique for calculating the total material inputs and environmental releases associated with the production, use and disposal of a product or service (Hendrickson et al., 1998; Lave et al., 1995). It is variously referred to as cradle-to-grave analysis and well-to-wheel analysis. LCA based estimates of GHG intensity can, in principle, be used to design policies that take into account leakage of emissions to unregulated activities and sectors. Although LCA is used to calculate a variety of different environmental burdens, it has an especially important role in addressing climate change in a global economy. While there exists a rich economic literature on policy choice to address global externalities and under-provision of global public goods, the focus of this literature is on comparing different traditional policy instruments such as emission fee, cap and trade and an emission intensity standards (Baumol and Oates, 1971; Fischer and Newell, 2008; Benneer and Stavins, 2007) and the risk of pollution leakage through relocation of domestic industries to locations abroad with lax environmental standards(Burniaux and Martins, 2000; Mattoo et al., 2009; Paltsev, 2001). This literature does not discuss life cycle based regulations. Regardless of the type of policy instrument, say, tax, emission standard or a technology mandate, LCA can play an important role in determining the magnitude or the stringency of a tax, an emission standard or mandate, since ignoring lifecycle impacts may cause sub-global policies to be counter-productive on a global scale. Even under an economy-wide policy, LCA has a role to play in calculating the environmental footprint of imported goods.

LCA typically yields estimates of input intensity (say, fossil fuel, electricity, or water) and/or

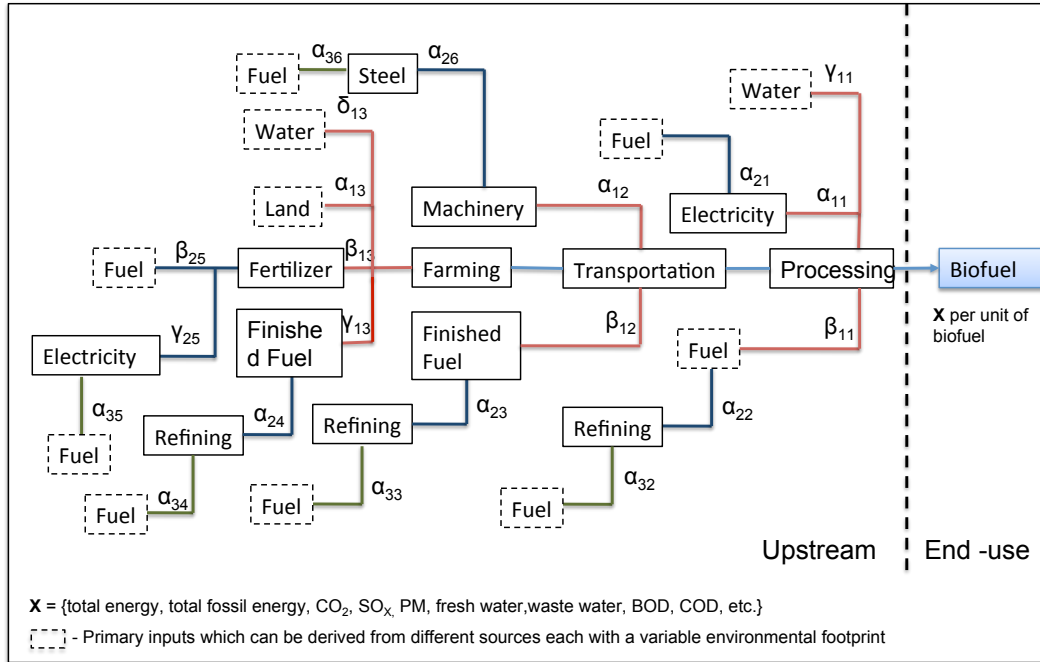


Figure 1: System boundary for a typical LCA of biofuel

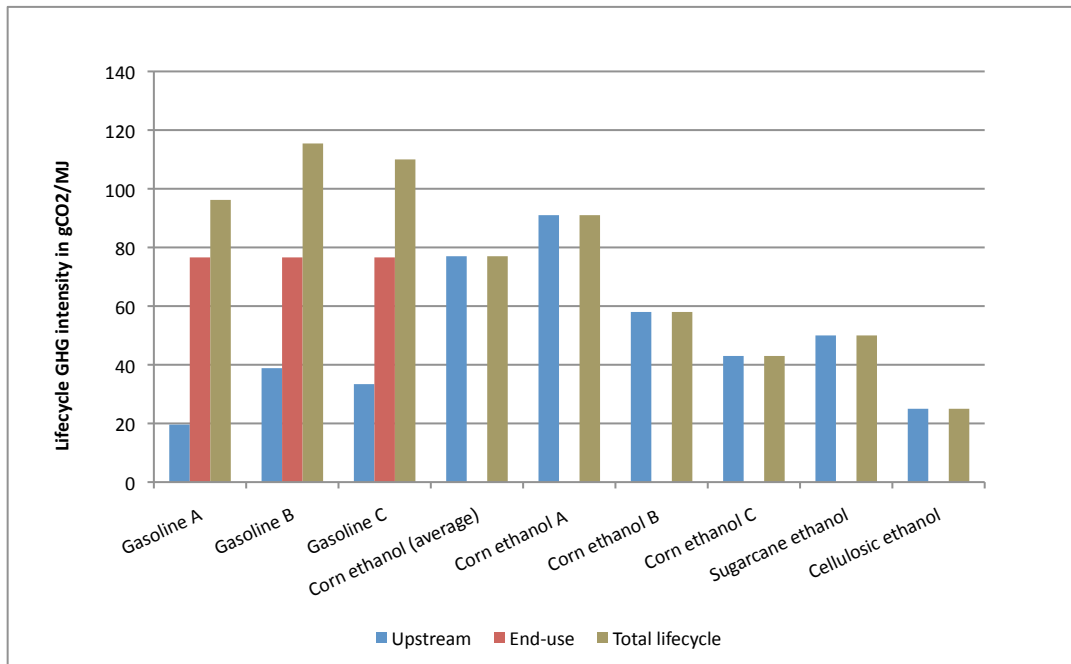
the pollution intensity (say, GHG, criteria pollutants, or toxics) of a final good (say, paper, cement, gasoline or biofuel) or service (say, air travel) (Hendrickson et al., 1998; Lave et al., 1995; Joshi, 2000). Figure 1 shows a typical system boundary for an LCA of biofuel. The dotted boxes represent primary inputs, such as fuel, land and water while the solid boxes intermediate products and final input. The quality of the primary input used at each step may differ in its pollution intensity. For instance, electricity production may be using coal, natural gas, hydro, nuclear or any combination thereof while fuel used for refining into liquid fuel may be derived from conventional crude oil or from non-conventional sources such as oilsands, gas/coal liquefaction.

To perform LCA, one requires information about the combination of inputs and technologies at each intermediate step in the lifecycle. In figure 1, the symbols $\alpha, \beta, \gamma, \delta$ represent the quantity of an input required to produce one unit of output. Depending on whether these input-output relationships represent that for a specific combination of processes or that for the average output of an entire industry and/or region, LCA can be used to calculate the environmental footprint of a specific firm or calculate the average footprint for a representative firm within an industry and/or region that produces a given good. One application of such LCA can be to compare the environmental footprint of different ways of producing a final good (say, corn ethanol produced by wet-milling Iowa corn using coal-based energy for processing versus corn ethanol produced by dry-milling Illinois corn using natural gas-based energy for processing). Another application of LCA can

be to compare the average environmental footprint of different final goods that may be substitutes (say, cane ethanol versus corn ethanol or ethanol versus gasoline). Farrell et al. (2008) through a meta-analysis of several different LCAs of corn ethanol showed that US corn ethanol has on average 20% lower GHG intensity than gasoline from conventional crude oil. They also found that the GHG benefits vary from less than 10% to more than 30% depending on the characteristics of the intermediate processes in the production of corn ethanol. Figure 2 shows representative estimates reported in the literature for the lifecycle GHG intensity in grams of CO₂/MJ of energy in corn ethanol. Similar assessments have been carried out for a wide variety of products and services such as comparison of asphalt and steel-reinforced concrete pavements (Horvath and Hendrickson, 1998), cement production (Huntzinger and Eatmon, 2009), electricity from different sources (Mann and Spath, 1999, 2001; Spath and Mann, 2000), automobile fuel/propulsion technologies (Lave et al., 2000; MacLean and Lave, 2003; Huo et al., 2009), paper versus polystyrene foam (Hocking, 1991), waste management (Craighill and Powell, 1996; Tillman et al., 1998; Moberg et al., 2005), different cropping systems (Kim and Dale, 2005; Tilman et al., 2006), different cellulosic biofuel technologies (Spatari et al., 2010) to name a few.

2 From *ex post* technology assessment to *ex ante* policy assessment

A survey by Cooper and Fava (Cooper and Fava, 2006) found that the traditional use of LCA is by firms for the purposes of supporting business strategy, research and development, product or process design, consumer education and environmental labeling. It is only recently that regulatory agencies have begun utilizing LCA to guide new environmental policies and to define targets. One of the first policies to adopt limits on lifecycle emissions is the US Renewable Fuel Standard (RFS) which was mandated under the Energy Security and Independence Act of 2007 (EPA, 2009a). The RFS stipulates an upper bound on lifecycle GHG intensity for biofuels that can be used to satisfy national biofuel targets. Another regulation is the California Low Carbon Fuel Standard (LCFS), which stipulates an upper bound on the lifecycle GHG intensity of the average transportation fuel sold within the State of California (ARB, 2009). Despite safeguards such as upper-bounds on lifecycle GHG intensity on biofuels, such policies are predicted to cause unintended negative environmental impacts (Delucchi, 2010; Rajagopal et al., 2010). A policy planner faces two major challenges in relying on existing LCA estimates for guiding policy selection, namely, scale and time effects and policy trade-offs. We describe these in detail.



Gasoline A: Gasoline from conventional crude
Gasoline B: Gasoline from oilsand
Gasoline C: Gasoline from natural gas
Corn ethanol (average): Average US corn ethanol
Corn ethanol A: Coal based biorefining, coal based fertilizers for farming
Corn ethanol B: Natural gas based biorefining, gas based fertilizers for farming
Corn ethanol C: Natural gas based biorefining + gas based fertilizers, 39% improvement in yield and 25% improvement in energy for processing
Cane ethanol: From Brazilian cane
Cellulosic ethanol: From Switchgrass (hypothetical)

Figure 2: Lifecycle GHG intensity of gasoline and ethanol from different sources and pathways (market-mediated indirect effects not included)

2.1 Scale and time effects

Whereas LCAs generally describe the lifecycle performance today, it provides little information about the long-term future environmental performance of a product. LCA provides limited information about how the environmental footprint of new technologies respond to scale, say a 100-fold increase from 10 million units to 1 billion units of a biofuel or wind energy or even a fossil fuel technology such as oilsands or deep, offshore oil drilling. The environmental or other benefits of switching one produce or process with its substitute, as suggested by a comparison of their LCAs, may not be captured on a large scale in the real world (Delucchi, 2005; Rajagopal and Zilberman, 2008b; Hillman and Sanden, 2008). Before we formally identify the structural reasons that explain the differences between LCA and assessments of future impacts based on economic models, we describe mathematically the assumptions implicit in relying on current environmental indicators to infer about future impact on emissions. For simplicity of mathematical exposition we depict one region and two technologies that are substitutes. However, the framework below can be extended to include an arbitrary number of goods in the economy and an arbitrary number of regions. Let Z denote GHG emissions, γ denote GHG emission intensity based on an LCA of the direct supply-chain and end-use, q denote quantity of fuel. Let the subscript t denote the time and the subscripts b and p denote a baseline scenario and a policy scenario respectively. Let the subscripts h and l denote a high and a low GHG intensity technology respectively. Let \tilde{Z} denote emissions from the rest of the economy. Then,

$$Z_{t_0} = \gamma_{t_0}^h q_{t_0}^h + \gamma_{t_0}^l q_{t_0}^l + \tilde{Z}_0 \quad (1)$$

$$Z_{bt_1} = \gamma_{bt_1}^h q_{bt_1}^h + \gamma_{bt_1}^l q_{bt_1}^l + \tilde{Z}_{bt_1} \quad (2)$$

$$Z_{pt_1} = \gamma_{pt_1}^h q_{pt_1}^h + \gamma_{pt_1}^l q_{pt_1}^l + \tilde{Z}_{pt_1} \quad (3)$$

The three equations above represent GHG emissions at present (t_0), at time t_1 under the baseline scenario and at time t_1 under the policy scenario respectively. The impact of the policy on GHG emissions at time t_1 is

$$\begin{aligned} \Delta Z_{pt_1} &= Z_{pt_1} - Z_{bt_1} \\ &= [\gamma_{pt_1}^h q_{pt_1}^h + \gamma_{pt_1}^l q_{pt_1}^l] - [\gamma_{bt_1}^h q_{bt_1}^h + \gamma_{bt_1}^l q_{bt_1}^l] + \tilde{Z}_{pt_1} - \tilde{Z}_{bt_1} \end{aligned} \quad (4)$$

The impact on emissions as suggested by a comparison of current LCA assuming one-to-one

replacement of high and low carbon technologies is,

$$\begin{aligned}\Delta Z_0^{lca} &= \left[\gamma_{t_0}^h (q_{t_0}^h - q_{pt_1}^l) + \gamma_{t_0}^l q_{pt_1}^l + \tilde{Z}_0 \right] - Z_{t_0} \\ &= \left[\gamma_{t_0}^h (q_{t_0}^h - q_{pt_1}^l) + \gamma_{t_0}^l q_{pt_1}^l \right] - \left[\gamma_{t_0}^h q_{t_0}^h + \gamma_{t_0}^l q_{t_0}^l \right] + \tilde{Z}_{pt_1} - \tilde{Z}_0\end{aligned}\quad (5)$$

Without loss of generality, let $q_{bt_1}^l = q_{t_0}^l = 0$ i.e, consumption of the low carbon technology, both at present and in the future baseline scenario is zero, i.e., the policy is binding. Then, the change in net emissions implied by comparison of current LCA would equal the change in emissions due the policy, i.e., $\Delta Z_0^{lca} = \Delta Z_{pt_1}$ when the following conditions hold

$$\gamma_{pt_1}^h = \gamma_{bt_1}^h = \gamma_{t_0}^h \quad (6)$$

$$\gamma_{pt_1}^l = \gamma_{bt_1}^l = \gamma_{t_0}^l \quad (7)$$

$$q_{bt_1}^h = q_{pt_1}^h + q_{pt_1}^l \quad (8)$$

$$\tilde{Z}_{bt_1} = \tilde{Z}_{pt_1} \quad (9)$$

Equations (6) and (7) imply that average emission intensity from the direct lifecycle, i.e., direct supply-chain and end-use, is constant. Equation (8) implies an assumption that total quantity consumed of the basket of substitute goods is unchanged under the baseline and the policy scenario. Finally, equation(9) implies the assumption that the emissions in the rest of the economy are unchanged between the baseline and policy scenarios. We discuss below reasons why these assumptions may be too strong to be satisfied and cite evidence from the literature in the context of ethanol.

1. **Variability in technical co-efficients in production** $\left(\frac{d\gamma^l}{dt} \neq 0, \frac{d\gamma^l}{dq^l} \neq 0 \right)$: This can be due to changes caused by several factors.

- (a) **Input substitution**: The technical relationship between inputs and output, which one observes either for the activity of a specific firm or for a representative firm within an industry, is not purely technical. It also embodies behavioral decisions, say profit-maximization by price-taking producers in a competitive industry. Under reasonable assumptions of limited substitutability in the short-to medium term and full substitutability in the long term between various inputs such as energy, capital and labor and between different types of a certain input such as energy from coal and energy for natural gas, a change in relative prices of different inputs will cause producers to adjust the optimal combination of inputs. The change in relative prices may be exogenous,

say, due to a tax on carbon which will increase the price of coal relative to natural gas as an energy source, or an endogenous change resulting from general-equilibrium effects caused by scaling up production of the final good whose environmental benefits is being assessed. Such adjustments will alter the average emission intensity of an industry and thus alter the lifecycle footprint of a final good. We illustrate this with an example. Farrell et al. (2006) through a meta-analysis of different LCAs of US corn ethanol, report that on average each liter of corn ethanol produced in the US displaces 0.18 kilograms of carbon di-oxide equivalent emissions. Their conclusion is based on the assumption that 90% of the biorefineries utilize natural gas and the remaining utilize coal. We performed sensitivity analysis of their model to various assumptions about the relative mix of coal and gas based energy input to fertilizer production and for processing of corn. The results are shown in table [1]. In the extreme case when both the biorefinery processes and fertilizer production utilize coal corn ethanol has only 5% GHG intensity per MJ relative to gasoline. On the other hand when both the biorefinery processes and fertilizer production utilize natural gas then corn ethanol has 43% lower GHG intensity per MJ.

- (b) Scale economies and technical change: Empirical evidence suggests that input-output relationship for infant industries tends to decline due to factors such as economies of scale (plant size) and learning-by-doing (Nemet, 2006) and due to technical change in the form of improvements in the quality of inputs (Hillman and Sanden, 2008), such as more efficient energy conversion technologies (Newell et al., 1999), better quality seeds (Evenson and Gollin, 2003), improved human-capital (Foster and Rosenzweig, 1996) etc. US Department of Agriculture (USDA) statistics suggest that corn yield per acre has been increasing at an average rate of about 1.7 percent per year between 1978 and 2008 (EPA, 2009b). Thus whereas Farrell et al. (2006) assume a corn yield of 140 bushels(bu)/acre, USDA predicts an average productivity of 160 bu/acre for 2010/11.² It is expected to reach 178 bu/acre by 2019/20, a cumulative increase of 27% compared the baseline. Table 1 shows the impact of increase in corn productivity and improvements in conversion efficiency.

2. Impact of aggregate final output on upstream and downstream industries ($\tilde{Z}_{bt_1} \neq \tilde{Z}_{pt_1}$):

- (a) Effects on input-producing sectors: Traditionally, LCA has placed emphasis on tracing activities along the vertical supply chain of a final product or service. For instance, the

²<http://usda.mannlib.cornell.edu/usda/ers/94005/2010/Table18.xls>

lifecycle of a finished fuel can be visualized as comprising of three main stages, namely, raw material extraction and transportation, refining and distribution, end-use combustion. Each stage may itself be further broken into its constituent vertical supply chain. Each type of material input (say, fossil energy, electricity, water) and pollution (say, GHG emissions, air pollutants, waste water) is aggregated across the multiple stages to determine the input and pollution intensity of the final good. Such vertical breakdown of the lifecycle ignores the horizontal structure of the sector producing the intermediate inputs in the supply chain. The horizontal inter-linkages can be a source of significant unintended impacts. For instance, the LCA of biofuel ignores the fact that the allocation of farmland to production of a biofuel crop competes for and displaces land from use for producing food, fodder, fiber and land for nature. Thus an increase in biofuel production forces an expansion of agriculture into non-farm lands, which has termed as Indirect Land Use Change (Searchinger et al., 2008; Melillo et al., 2009; Dumortier et al., 2009; Havlik et al., 2010; Hertel et al., 2010). Figure 3 below illustrates one possible chain of linkages leading to ILUC due to a US policy that mandates increase in corn ethanol consumption. ILUC emissions differ for different biofuel pathways. A simulation of the Food and Agricultural Policy Research Institutes model of international trade in agricultural commodities predicts that whereas a 1% increase in US corn ethanol production leads to 0.009% increase in world crop area, a 1% in Brazilian sugarcane ethanol production leads to only 0.001% increase in world crop area (Fabiosa et al., 2009). In other words, the ILUC impact of a unit of biofuel from Brazilian cane is 1/9th that for biofuel from corn produced in the US. (See table 1)

Another form of pollution leakage is *pollution shuffling*. When an intermediate input is homogenous in functionality but heterogeneous in its environmental footprint, say electricity produced with coal versus electricity produced with wind turbines, LCA based regulations can lead to outcomes wherein cleaner inputs are allocated to the regulated activities, say fuel production in US and polluting inputs are allocated to the unregulated regions, say fuel production outside of US or unregulated sectors, say food production. Rajagopal et al. (2010) argue that regional policies such as California's LCFS may reduce consumption of oilsands within California and increase their consumption elsewhere.

- (b) Effects on end-use sector ($q_{bt_1}^h \neq q_{pt_1}^h + q_{pt_1}^l$): In extrapolating LCA-indicators to infer about total change in emissions, it is often implicitly assumed that a given quantity of one good, say low carbon fuel, simply replaces an equal (or equivalent amount if one has

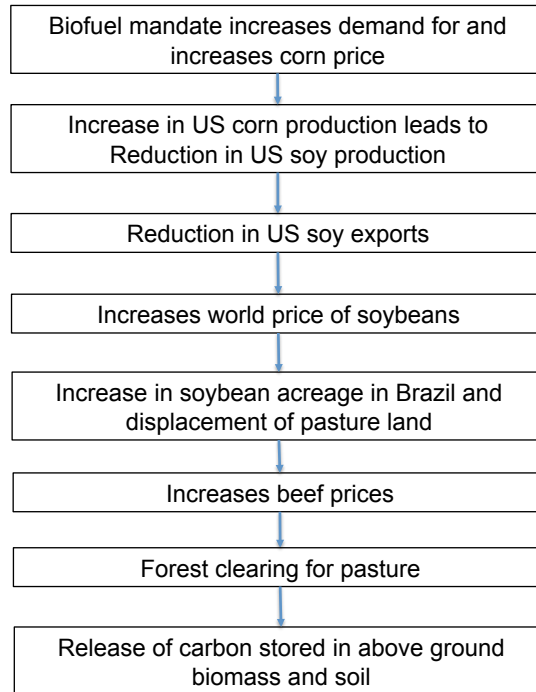


Figure 3: Illustration of one possible chain events leading to indirect land use change emissions

to normalize the two goods to a common unit) of a dirtier substitute. In other words it is assumed that total consumption remains unchanged. However, often the introduction of the clean good, affects the price of the good and its substitutes, thereby affecting total consumption (Rajagopal et al., 2010). For instance, the introduction of biofuels in the U.S. has been shown to reduce the world price of oil and cause a rebound in world oil consumption. In other words, 10 billion (gasoline-equivalent) gallons of ethanol will replace less than 10 billion gallons of gasoline because of the rebound effect. Terming this effect as indirect fuel use change (IFUC), Rajagopal et al. (2010) show IFUC that can significantly attenuate or amplify the direct supply chain emissions. In fact, they show that depending on the policy regime, IFUC can counter ILUC and reduce the effective lifecycle GHG intensity of biofuels (See table 1). They also argue that IFUC effect due to biofuels differs for different policy regimes. When the effect of introduction of the new technology is to reduce the cost of the final good then the IFUC effect may amplify lifecycle emissions.

3. **Emission allocation under joint production:** Production processes often yield multiple products. For instance, corn ethanol is jointly produced with distillers grains (DG), a substitute to raw corn grain as feed for livestock operations. Similarly, ethanol production from cane yields bagasse, a fibrous residue, that is combusted to captive generation of heat and/or

power. To give a non-biofuel example, the distillation of crude oil yields multiple petrochemicals including gasoline, diesel, jet fuel, naphtha, coke etc. The allocation of lifecycle emissions across multiple products reduces the environmental footprint of any one product. As production expands the demand for co-products may change warranting a corresponding adjustment in co-product credit. Assuming DG is a perfect substitute for corn grain and since one-third the quantity of raw corn grain results as DG along with ethanol, current LCA's of biofuel allocate only two-third the total emissions from corn processing to ethanol (Wang, 1999; Farrell et al., 2006; Liska et al., 2008). When compared with the fact, the direct lifecycle GHG intensity of corn ethanol is only 20% lower than gasoline, the importance of co-product credits becomes apparent. (See Rajagopal and Zilberman (2008a) for a discussion of the different approaches of co-product allocation).

4. **Counterfactual assumption** $\left(\frac{d\gamma^h}{dt} \neq 0, \frac{d\gamma^h}{dq^h} \neq 0\right)$: The net benefit (or cost) of one technology depends on the choice of the baseline relative to which the net impact is calculated. The International Energy Agency predicts that more than 75% of the increase in demand for liquid fuel between 2006 and 2030 will be met through liquids from unconventional sources such as oil sands and liquefaction of natural gas and coal whose lifecycle GHG intensity is higher than conventional crude oil (IEA, 2008). Holding input-output relationships of renewable fixed, an increase in the relative GHG intensity of fossil fuel increases the relative benefits of non-fossil fuel technologies. On the other hand if one assumes a baseline involving carbon cap and trade program in the US and/or a global agreement on climate change, the relative benefits of renewable energy decreases for the counterfactual now involves slower transition to non-conventional fossil fuels. See table 1 for sensitivity of benefits of ethanol to 10% increase in GHG intensity of oilsand.

5. **Planning horizon and discount rate**: Both decisions concerning private investments in capital intensive activities and public policy selection involve a planning horizon spanning several years or decades and the profile of emissions may vary over time. Similar to the notion of a financial pay back period beyond which investments deliver net positive benefits, the pay back period beyond which net lifecycle emission benefits begin to accrue for those investments will be positive. For biofuels, the pay back period may be in the range of decades (Fargione et al., 2008; Havlik et al., 2010). Different methods and rates of discounting GHG emissions occurring at various points time from land use change will result in widely varying estimates of the environmental benefits of biofuel (EPA, 2009b; Dasgupta, 2008). O'Hare et al. (2009) show that calculations of simple payback period beyondfor GHG emission from

land conversion to be compensated by the GHG benefits of switching from fossil fuels to biofuels undervalue the environmental cost of land use change emissions relative to the value of benefits from fuel switching. Mathematically, this implies,

$$\int_{t_1}^{t_2} e^{-\rho t} dD(Z_{p_1 t}) \neq \int_{t_1}^{t_2} e^{-\rho t} dD(Z_{p_2 t}) \quad (10)$$

where, p_1 and p_2 are two different policy scenarios, $D(\cdot)$ is the net damage (or benefit) as a function of emissions and ρ is the discount rate

In economic terminology, the reasons we identify above, with the exception of the issue of inter-temporal distribution of net benefits, can collectively be described as general equilibrium effects. Indirect land use change and indirect fuel use change are manifestations of general equilibrium effects when analyzed within a partial equilibrium context. In a competitive industry, general equilibrium effects are not caused by the actions of any single agent but are an emergent property of the system. Therefore, no single biofuel producer causes land use change but the aggregate impact of the actions of all agricultural producers and consumers affects the market outcome. When there are no missing markets i.e., either when environmental externalities are absent or when there exists a globally consistent policy regime in the form of a global emission tax or a global market for emission permits, then general equilibrium effects are merely pecuniary externalities that do not affect the efficiency of resource allocation but affect income distribution. In such cases, general equilibrium effects on lifecycle emissions are not relevant from policy standpoint. Such conditions also obviate the need for lifecycle accounting altogether leave alone accounting of general equilibrium effects. However, the reality is that pecuniary externalities lead to impecuniary externalities, which are policy relevant.

There exist two approaches to computing market-mediated emissions in the context of biofuels. One approach is to compute the indirect effect within each sector and summing the various indirect effects across all sectors of interest to compute the average aggregate indirect emission intensity per unit of output. This is then added to the direct lifecycle GHG intensity based on emissions from direct supply-chain and during end-use. This is approach for computing adopted by the regulatory agency, California Air Resources Board in implementation of the Low Carbon Fuel Standard (ARB, 2009). An alternative approach is one that does not categorize emissions into direct and (different) indirect emissions but instead computes the change in total emissions between two future states of nature, one of which represent the outcome of a new policy action and the other a business-as-usual baseline in which current policy regime is maintained in the future. This is the approach adopted by the US Environmental Protection Agency in implementation of the national Renewable Fuel

Table 1: Sensitivity of lifecycle GHG benefits from corn ethanol

	change in GHG intensity relative to baseline ethanol ^a	GHG intensity relative to gasoline ^b
Baseline corn ethanol Farrell et al. (2006)	na	-18%
<i>Input substitution</i>		
Biorefinery processes and fertilizer production utilize only coal ^c	11%	-9%
Biorefinery processes and fertilizer production utilize only natural gas ^c	-30%	-43%
<i>Technical change</i>		
15% increase in corn yield from 140 bu/acre to 160 bu/acre ^d	-6%	-23%
10% increase in conversion efficiency from 2.7 gallons per bushel to 3 gallons per bushel ^e	-4%	-21%
<i>Emission allocation under joint production</i>		
25% reduction in co-product credit	4%	-15%
25% increase in co-product credit	-12%	-28%
<i>Impact on input producing sector - ILUC</i>		
Hertel et al. (2010) ILUC estimate of 30 gCO ₂ /MJ ^f	39%	14%
<i>Impact on fuel consumption</i>		
Rajagopal et al. (2010) IFUC estimate for renewable Fuel Standard (for their medium elasticity scenario)	-32%	-48%
<i>Counterfactual assumption - Baseline</i>		
10% increase in GHG intensity of oil ^g	na	-26%
<i>Discounting</i>		
30 year investment horizon and 0% discount ^h		5%
100 year investment horizon and 2% discount ^h		-16%

^a Baseline ethanol GHG intensity is 77 gCO₂/MJ (Farrell et al. (2006))

^b Baseline gasoline GHG intensity is 94 gCO₂/MJ (Farrell et al. (2006))

^c We modified the assumptions in the EBAMM model of Farrell et al. (2006)

^d <http://usda.mannlib.cornell.edu/usda/ers/94005/2010/Table18.xls>

^e See page 12, table 1.1-1 of EPA (2009b)

^f There is a large literature with wide ranging estimates for payback period and average ILUC emissions per unit of corn ethanol. While Searchinger et al. (2008) calculate 107 gCO₂/MJ, Tyner et al. (2010) predict a range between 13.9 and 22.9 gCO₂/MJ. Since Searchinger et al.'s estimate is considered too pessimistic, we chose Hertel et al. (2010)'s estimate for illustration here.

^g See Charpentier et al. (2009) for a meta-analysis of LCA of oilsands

^h See EPA (2009a) Notice of Proposed Rulemaking for Table VI.C.1-1. This report compares future corn ethanol to a fixed 2005 petroleum baseline and so underestimates the net GHG benefits

Standard (EPA, 2009b). However, both the regulations focus only on quantifying indirect land use impacts. The method for treatment of inter-temporal effects is also a matter of debate.

2.2 Policy trade-offs

The policy-selection problem can be formulated as a decision problem that involves choosing a policy instrument or multiple instrument to achieve one or more objectives while satisfying certain constraints. Policy makers have at their disposal a wide array of instruments to choose from including pollution tax, subsidy, emission cap, performance standard (say, emission intensity standard or energy efficiency standard), technology standard and mandates, labeling and information disclosure etc. Lifecycle assessment is most relevant when policy makers selectively favor certain technologies through a subsidy for a new technology or through regulations such as technology standard (say, mandatory installation of scrubbers in electric power plants or catalytic converters in automobiles), and renewable electricity/fuel standards. In contrast, pricing pollution (either directly through a tax or indirectly through a system of tradable pollution permits subject to a cap) is a technology-neutral policy. A price on pollution equal to the marginal social cost of pollution maximizes welfare. In reality, either uncertainty in marginal social cost or political economic constraints render first-best policies infeasible. Emission intensity standards and energy efficiency standards can in theory be technology-neutral but in practice the stringency of the standard may be dictated by the standard of available alternatives, in which case they are *de facto* technology standards. An example is the California Low Carbon Fuel Standard where it is expected that biofuels will be the principal mechanism by which the GHG intensity targets will be achieved. It is hence equivalent to a renewable fuel standard. However, a difference between a performance standard and a technology standard/mandate is that former leaves the door open for new technologies in the future. For instance, future breakthrough in the cost of storage technology may render electric propulsion more cost-effective than biofuels.

The difficulty in interpreting LCA for picking technical winners is that different technologies may present different trade-offs between various environmental burdens. A technology that reduces lifecycle GHG emissions may yet increase regional air pollution or other environmental burdens. Jacobson (2009) finds that both corn-E85 (85% ethanol and 15% gasoline) and cellulosic-E85 degrade air quality by up to two orders of magnitude more compared to electric propulsion systems that are powered by electricity from sources such as solar and wind energy. A study by the Organization for Economic Cooperation and Development (OECD, 2008) compares three different environmental benefits, namely, climate change, surface water quality and biodiversity for three different types of biofuels produced in the EU, namely, cellulosic ethanol from Reed Canary Grass,

ethanol from wheat and biodiesel from rapeseed. The indicators used to measure the three benefits are lifecycle CO₂ emissions, nutrient runoffs from fertilizer (N and P) and herbicide use, and a habitat index respectively. The habit index in turn relies on quality of habitat that a biofuel plantation provides for butterflies. They find that second-generation biofuels from reed canary grass has the least lifecycle CO₂ emissions. Wheat performs poorly with respect to both CO₂-eq emissions and nutrient runoff because of high fertilizer and herbicide use intensity. However, rapeseed has the highest biodiversity benefit per unit of land because it affords the best habitat for butterflies.

Furthermore the objective function of a policy maker often may include other objectives in addition to reducing pollution. For instance the US Energy Security and Independence Act 2007 states at least four different justifications are cited for renewable energy policies, namely, reducing environmental pollution, improving energy security, reducing consumer cost of energy, and generating employment.³ While our emphasis has been on GHG accounting, different technologies may involve different trade-offs between different objectives. For instance, the GHG intensity of the direct supply chain (ignoring indirect non-supply chain effects) for Brazilian cane ethanol is on average 30% lower than that for corn ethanol produced domestically within the US. However, cane ethanol offers lower energy security and domestic employment benefits compared to corn ethanol. Unlike other renewable and fossil sources of energy, biofuels involve a unique trade-off between food and fuel. Biofuels policies have had a negative impact on food consumers, especially the poor in developing countries and an important contributor to the global food crisis in 2008 (Abbott et al., 2008; FAO, 2008; Hochman et al., 2010b). The second-generation of biofuels from cellulosic biomass are expected to mitigate these trade-offs.

The fact that such trade-offs exist is not a criticism of LCA. LCA is a general framework for holistic assessment of the environmental burden of a technology across multiple dimensions such as GHG emissions, criteria air pollutants, toxics, fresh water, waste water, net energy balance, fossil energy use, electricity etc. The principle of policy targeting suggests that when there are multiple policy targets, the number of policy instruments should at a minimum equal the number of policy targets (Tinbergen, 1952). For instance, if policy makers care about addressing both climate change and regional air pollution then they should attach a separate policy instrument to each externality and not rely on a single policy to target both. Similarly, if the goal is to address climate change and address under-investment in R&D, the optimal outcome is achieved through a price on GHG emissions and a subsidy for R&D Fischer and Newell (2008). The goal of our discussion is only to highlight the need for integrating LCA with a framework such as welfare

³<http://www.gpo.gov/fdsys/pkg/PLAW-110publ140/content-detail.html>

analysis or cost-effectiveness analysis that can be used for comparison of different technological possibilities.

3 On incorporating markets and policies with LCA

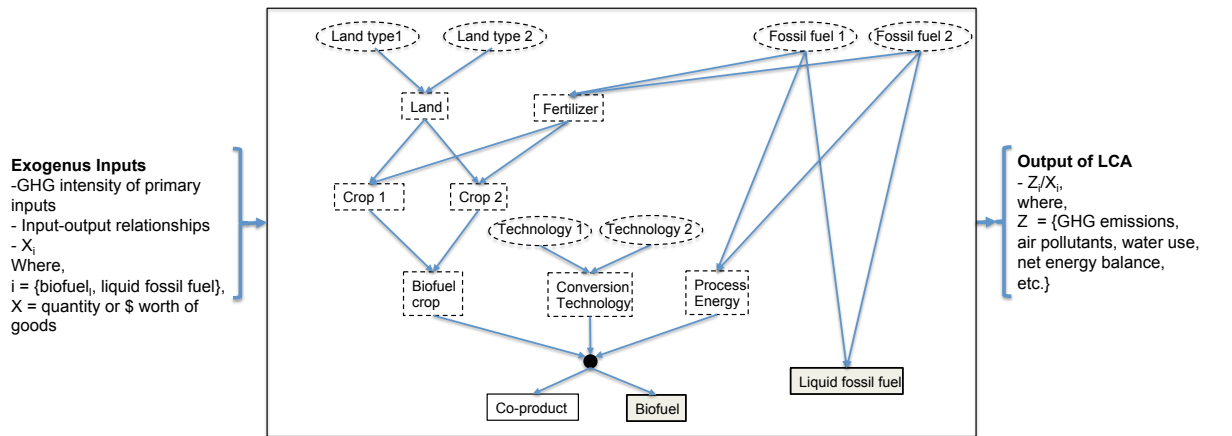
Having highlighted the importance of economic variables and policy parameters in LCA, we briefly highlight the strengths and limitations of alternative approaches for incorporating markets and policies. Among the existing LCA techniques, Economic Input-Output (EIO) LCA, can compute both the supply-chain and the non-supply chain effects simultaneously since an EIO table contains information about the flows of goods and services between the various sectors of an economy. However, it implies both fixed input-output relationship in production and fixed final consumption when predicting the future. The level of aggregation of activities within an IO table may make it unsuitable for analyzing emerging technologies such as alternative energy and non-conventional fossil fuels that comprise a small portion of the aggregate output of a large sector. This is possible using a Process LCA approach. However, process LCA does not model the indirect emissions. While a hybrid approach combining EIO LCA and process LCA, can overcome the limitation of aggregation and address indirect effects, both approaches impose the assumption of fixed proportion production function and fixed consumption. This suggests that input-output (IO) accounting is appropriate for calculating the present environmental footprint of a technology or at a time in the past. Relying on current techniques, one is forced either to assume that the technical input-output relationships (and therefore the environmental performance) are invariant to scale or to make explicit assumptions about how the technical relationships will change. The former is inadequate (as evident from the debate about the land use change due to biofuel expansion and its impacts) while the latter is challenging (especially for infant technologies which are evolving rapidly and involve scale effects). Secondly, policy makers choose policies to target multiple objectives such as pollution reduction, energy security and reducing consumer cost while satisfying certain constraints. In such cases, LCA provides little guidance about the trade-offs between different technology options.

A more flexible but also a more complex alternative to IO framework is a computable general equilibrium (CGE) framework (West, 1995; Rose, 1995). CGE models analyze economy-wide impacts of a 'shock' including inter-sectoral impacts, effects on trade patterns and changes in factor prices. CGE models have a long and rich history of application in policy analysis (Dixon and Parmenter, 1996). It relaxes the assumption of fixed-proportion production and fixed consumption and contains explicit supply constraints, all embedded in a neoclassical framework. In contrast

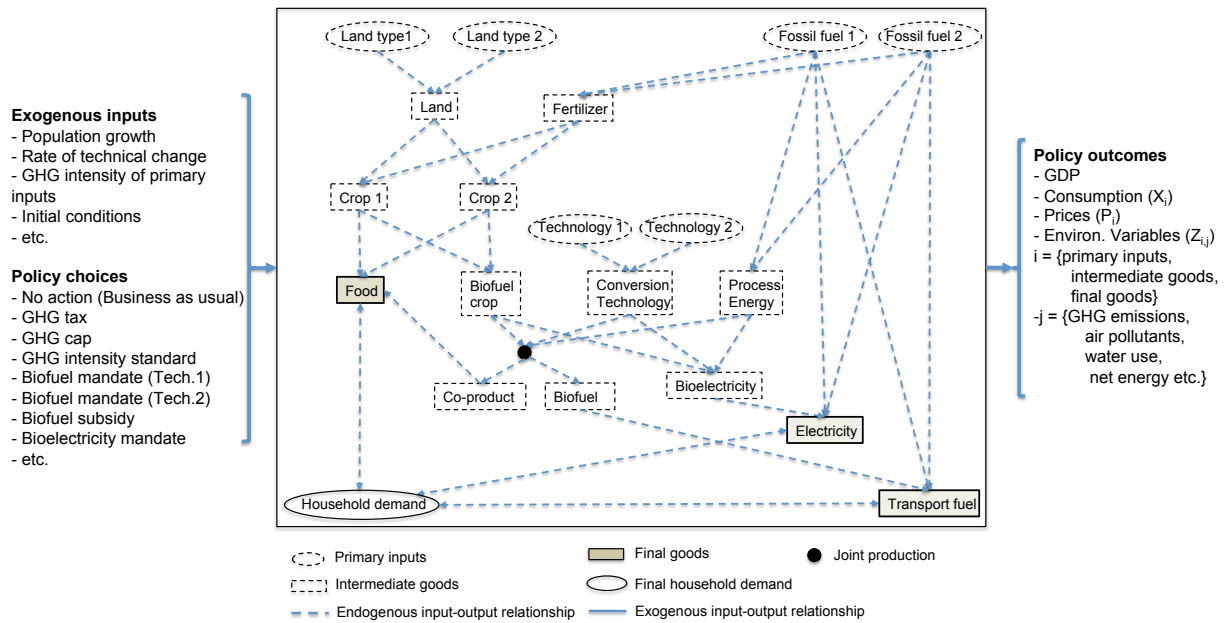
to IO, CGE model is an optimization model, i.e. it reallocates resources optimally in response to an exogenous shock which can represent a sudden technical change (e.g. breakthrough in cellulosic technology), a policy shock (e.g. a carbon tax or a mandate) or a pure economic shock (e.g. weather shock or supply embargo). Figure 4 depicts some of the differences between lifecycle assessment and general equilibrium analysis of a policy or economic shock. For illustrative purposes and without loss of generality we assume that consumers demand three final goods, namely, food, transport fuel and electricity, and there are two primary inputs, namely, land and fossil fuels which are combined to produce the intermediate goods and final goods, and that each of the two primary inputs can be derived two different sources which differ in their GHG intensity.⁴ Land and fertilizer, an intermediate input, can be utilized to produce two types of crops each of which can be used for food or biomass for energy production. Biomass can be converted using energy and one of two conversion technologies into biofuel or bioelectricity which can substitute liquid fossil fuel and fossil-based electricity respectively. Whereas LCA assumes an exogenous input-output relationships (represented as solid lines in panel 4(a)), the input-output relationships are an endogenous variable in an equilibrium framework which only assumes a production function (represented as dotted lines in panel 4(b)). We can see that an advantage of equilibrium analysis is it incorporates the competition for resources at each stage of the lifecycle such as the competition of land for food and fuel, the competition of fossil fuel for food, fuel and electricity and the competition of biomass for fuel and electricity etc. The equilibrium framework also endogenously determines the total quantity of goods produced which is an exogenous input for LCA. An advantage of LCA relative to economic equilibrium models is that it incorporates significantly more detail in the supply chain of various end products or services for which it is applied. A comparison of figure 4(b) with figure 1, which shows the breakdown in typical LCA, makes this apparent.

In theory, a general equilibrium analysis can fill the gaps we identified earlier in using LCA for deriving policy conclusion. In practice, the CGE approach has its own limitations. Being a model of the entire national or global economy CGE models are computationally complex. This limitation is however being overcome by the advent of faster computers. A more serious limitation pertains to availability of disaggregated micro-level data on production and consumption. The lack of such data inhibits the specification of sophisticated models of production and consumer behavior forcing neoclassical specifications which may not be a good representation of reality. Even for simple specifications, empirical data tends to be outdated or simply absent forcing modelers to make assumptions that are hard to validate. Sensitivity analysis to assumed parameters tends to

⁴Both LCA and general equilibrium frameworks can be extended to include other primary inputs such as capital and labor and an arbitrary number of different types of each input (including other renewable and non-renewable inputs), final goods, production technologies, and intermediate stages in the supply chain of final goods.



(a) Lifecycle assessment framework



(b) Market equilibrium framework

Figure 4: Illustrative comparison of lifecycle assessment and general equilibrium analysis

be complicated when the number of such parameters ranges in the tens or even hundreds.

If only a limited number of inter-industry interactions are important in the lifecycle, and if macro-economic linkages have only second-order importance, then a CGE approach would be inefficient. Far too much time and effort will have to be spent in calibrating and validating aspects of the model that are not of critical importance. A simpler approach is a multi-sector, partial equilibrium model focussing on linkages between a small number of markets, which are strongly linked either on the supply side or the demand side (Sadoulet and De Janvry, 1995). By focussing on a small number of sectors, multimarket models allow for a richer specification of the technology and market structure in the relevant sectors. Figure 4(b) can be amended to represent a multi-market partial equilibrium by making income exogenous (and correspondingly relaxing one of the equilibrium closures). It is worth mentioning that the US Environmental protection agency (EPA) has adopted a multi-market partial equilibrium approach to compute the indirect effects of biofuel policies. However it focuses solely land use change impacts and ignores other types of effects we mention above.

4 Conclusion

One economic rationale for introducing LCA-based policies is that the environmental impact of a commodity like fuel is the outcome of activities that are spread across many sectors and across different countries some of which may not be subject GHG regulations. Actions to reduce emissions from a subset of activities can inadvertently increase global emissions. LCA can help identify by policy actions that are not counter-productive. The logic of LCA-based regulation is that one agent, say, the seller of a final good such as transport fuel, is responsible for emissions occurring at various stages of the supply chain. In contrast, traditional regulation is based on the principle that the polluter is responsible only for his own emissions. A potential advantage of LCA from a policymakers standpoint is that by shifting responsibility for the GHG emissions of the entire supply chain to the seller of the fuel it reduces transaction and monitoring costs. LCA is a proven, empirically-based technique for ex post assessment or for ex ante assessment of small shocks. Policy planners ought to exercise caution when using this information for making long-term policies. The effects of scale and time, which are important for long-term policy planning, have not received sufficient attention in the literature so far. The debate about the lifecycle impact of biofuels is a case in point. Policy makers need to clearly distinguish between the current benefits revealed by a comparison of the current supply-chain of different technologies (or products or services) that are substitutes and the likely future benefits of replacing one technology with another on a large

scale. Furthermore, different combinations of technologies and policies involve different trade-offs between the various environmental and socio-economic indicators. An integrated assessment of technology, markets and policies can lead to better policies.

In addition to the methodological issues in accounting arising from market-mediated economic feedbacks, we recognize that technical understanding of certain types of processes in the supply-chain itself or of their impacts even if we can account the physical flows, may be limited. For instance, quantifying the impacts of biogeophysical and of ecological processes that occur over a long time scales is inherently more complex compared to quantifying the flow of energy and materials through industrial processes. Therefore the impacts of application of Nitrogen fertilizer and mineralization of manure from livestock, the impact of land-use change on soil carbon vary with ambient conditions, soil management practices etc. are fundamentally uncertain (Delucchi, 2010). These are important topics for future research. A direction for future research is the development of a framework that combines the strengths of different modeling techniques such as traditional LCA, biophysical models of land and land use change, and economic equilibrium models and is parsimonious with regard to data requirements. Such a framework can serve as useful screening tool for selecting cost-effective technologies and policies whose performance is robust to a broad range of future economic shocks. We are aware this is easier than done. Our intention in this paper is not present a new approach but to highlight the structural differences between supply-chain focussed LCA and policy-focussed LCA.

Biofuels have been mixed bag for researchers and the field of LCA. While they have brought LCA to the forefront of policy discussion, they have identified a rich agenda for future research. It highlights the need for a framework to perform LCA of policies different from LCA of technologies. That said, existing frameworks of LCA may be more readily applied for designing environmental regulations in other sectors of the economy. Global commodities such as liquid fuels present special challenges for LCA-based regulations. LCA of biofuel is particularly complex, because its supply chain involves agriculture, which in addition to being a globalized activity is also diffuse and heterogenous source of emissions and therefore more uncertain, when compared to industrial production. Identifying the sectors of an economy that are more easily amenable to lifecycle-based regulations in a simple and transparent manner is a topic for future research.

The focus in this paper has been on the technical aspects of conducting LCA for policy purposes. Methodological challenges in calculation of lifecycle performance notwithstanding, the actual implementation of a regulation on lifecycle emissions faces additional difficulties. The traditional principle of environmental policy making has been that polluters are responsible for their own actions. However LCA-based regulations assign to one agent the responsibility for emissions caused

by others. Assigning responsibility for emissions due to general equilibrium feedbacks, especially non-supply chain effects, to any one agent is especially without precedent. It is worth reiterating that in competitive industry non-supply chain effects are not caused by any single agent but are an emergent property of the market system. Therefore, holding one sector, say the biofuel industry, responsible for actions over which they do not exercise any control (e.g. US biofuel producers do not control the behavior of land owners in Brazil who may decide to deforest the Amazon) poses legal and ethical issues. At the same time ignoring non-supply chain effects can be counter-productive to the policy goals. Resolving the inconsistency between micro-level, firm-specific LCA and macro-level LCA is crucial for implementing LCA based regulations.

Finally, given the informational and administrative burden on policy makers for designing effective partial policies for global problems, one cannot overstate the importance of achieving a binding global agreement on addressing climate change and the importance of implementing economy-wide policies that target emission reduction at source.

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