Title
Life Cycle Assessment for Economists

Journal
Annual Review of Resource Economics, 9(1)

ISSN
1941-1340

Authors
Rajagopal, Deepak
Vanderghem, Caroline
MacLean, Heather L

Publication Date
2017-10-05

DOI
10.1146/annurev-resource-100815-095513

Peer reviewed

Life Cycle Assessment for Economists

Deepak Rajagopal,[1,*] Caroline Vanderghem,[2] and Heather L. MacLean[2,3,4]

[1]Institute of the Environment and Sustainability and Department of Urban Planning, University of California, Los Angeles, California 90095; email: rdeepak@ioes.ucla.edu

[2]Department of Chemical Engineering and Applied Chemistry, University of Toronto, Toronto, Ontario M5S 3E5, Canada; email: c.vanderghem@utoronto.ca

[3]Department of Civil Engineering, University of Toronto, Toronto, Ontario M5S 1A4, Canada; email: heatherl.maclean@utoronto.ca

[4]School of Public Policy and Governance, University of Toronto, Toronto, Ontario M5S 3K9, Canada

Annu. Rev. Resour. Econ. 2017. 9:15.1–15.21

The *Annual Review of Resource Economics* is online at resource.annualreviews.org

https://doi.org/10.1146/annurev-resource-100815-095513

*Corresponding author

JEL codes: Q40, Q50, Q53, Q56, Q57, Q58

Keywords

life cycle, regulation, leakage, transaction cost, adoption, labeling, uncertainty

Abstract

Life cycle assessment (LCA) is a widely utilized technique to quantify inputs and emissions associated with the life cycle of a product, from raw materials extraction through the product's end-of-life. Given the basic economic principle of policy targeting, the case for focusing on emissions associated with a specific good as opposed to targeting each different externality needs development. This review identifies situations that merit a product life cycle approach in environmental regulation and then discusses the use of LCA with different types of policy instruments. We then discuss the methodological and implementation-related issues involved with using LCA as an economic decision aid as well as issues in designing regulations to control life cycle emissions. We conclude by identifying areas for future LCA research that are ripe for the application of microeconomic insights.

1. INTRODUCTION

Life cycle emissions refer to all emissions either directly or indirectly associated with the consumption (or production) of a specific good, activity, or service. For instance, the direct emissions from crop production are those occurring at the farm, whereas indirect emissions are those associated with the production of farm inputs. Emissions during the production and transportation of seeds, chemicals, fuels, and water are indirect emissions of the first order. Emissions during the production and transportation of inputs used to produce the inputs used on the farm are indirect emissions of the second order.[1] The sum of direct and indirect emissions is the life cycle emissions associated with the crop and coproducts (**Figure 1**).

Product life cycle assessment (LCA) is a widely utilized approach for quantifying the life cycle emissions as well as the resource inputs (e.g., different types of energy/fuels, water, and materials) associated with a product or service. The International Organization for Standardization (ISO) provides guidance on LCA and defines it as the "compilation and evaluation of inputs, outputs and the potential environmental impacts of a product system throughout its life cycle" (ISO 2006a, p. 2). LCA is often the foundation of bottom-up systems approaches. It was designed to examine environmental impacts of historical or current production, over short defined time periods, with a focus on identification of the biggest impact reduction potential and improvement strategies without burden shifting. LCA was established primarily to guide product development as well as to inform policies. LCA has been utilized for several decades in various industries (e.g., automotive, consumer goods) and in government [e.g., the US Environmental Protection Agency's (EPA) Resource and Environmental Profile Analysis (Hunt et al. 1992)]. More recently, LCA has been incorporated into a number of policies/regulations, primarily those related to reducing the

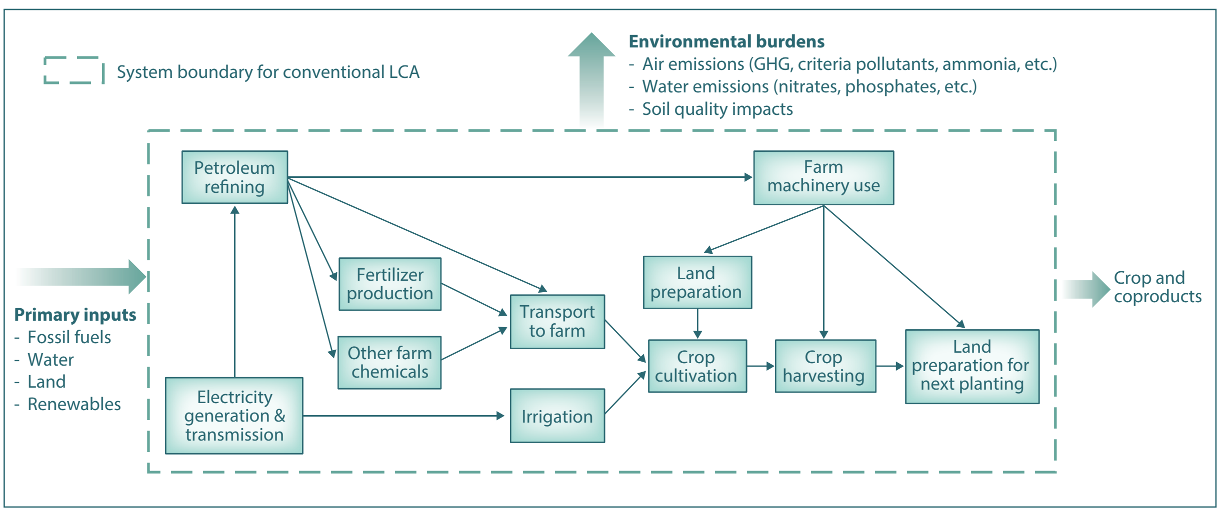

Figure 1

Schematic of a representative system boundary for life cycle analysis (LCA) of crop production. LCA involves accounting, and when appropriate, aggregating the material flows relevant to any given type of environmental externality of interest occurring within the chosen system boundary.

[1]Although one can continue to regress infinitely, depending on the type of LCA, either this is not an issue, or alternatively, heuristic stopping criteria could be employed. For instance, indirect emissions beyond the second order might be ignored.

greenhouse gas (GHG) emissions intensity of transportation fuels; regulations include the US Renewable Fuel Standard (RFS; **https://www.epa.gov/renewable-fuel-standard-program**) and the California Low Carbon Fuel Standard (LCFS; **https://www.arb.ca.gov/fuels/lcfs/lcfs.htm**). LCA has also played a key role in informing funding agencies and investors regarding the attractiveness of novel technologies (**https://www.sdtc.ca/en/about-sdtc/reports**). There are, however, several challenges in conducting LCA from an economic decision-making perspective. Although the field of LCA has expanded rapidly, many methodological and implementation-related challenges remain (e.g., Hellweg & Milà i Canals 2014, Ness et al. 2007, Reap et al. 2008), which are critical to understanding the role(s) that LCA could and should play in public decisions about energy and the environment.

The objectives of this review are the following. First, we establish the economic case for the use of LCA in reducing environmental externalities (Section 2). On a related note, we also outline the different ways in which the estimates derived from an LCA could be used with different policy instruments (Section 3). We then summarize the practical challenges in generating and employing estimates from LCA in a policy context (Section 4). We close the review by highlighting areas for future interdisciplinary research that requires collaboration between economists, engineers, policy analysts, and LCA practitioners.

2. THE RATIONALE FOR POLICY USE OF PRODUCT LIFE CYCLE INFORMATION

Environmental externalities provide a necessary, albeit insufficient, justification for public policies to target life cycle emissions traceable to a specific good or activity. The cost-effective way to internalize an externality is one that targets all actions contributing to that externality rather than a subset of such actions, which the consumption of that specific good typically represents. For instance, GHG emissions arise from fossil fuel use and land use in general, and fossil fuel and land uses are ubiquitous in the global economy. Likewise, when there is more than one type of externality of interest, the simplest approach is to attach a separate policy to each type of externality, which is the well-established Tinbergen principle of policy targeting (Tinbergen 1952). In this simple, neoclassical view of environmental regulations, the role for product life cycle information is limited (Portney 1993). However, while cautioning against any broad and mandatory policy use of product life cycle information, Portney (2012, p. 41) asserts that "the careful application of LCA can identify environmental effects that might otherwise have been ignored so that they can be included in a thorough and comprehensive BCA (benefit cost analysis)."

Here, we point out four real-world situations that merit an assessment of life cycle pollution associated with any given product or activity. In doing so, our goal is not to establish that LCA is a means of designing a single instrument to solve multiple environmental externalities, which is the main criticism of Walls & Palmer (2001) about LCA. Instead, we emphasize the policy relevance of the concept of product life cycle footprint for addressing any given externality, such as global warming, ocean acidification, or ozone layer depletion. The utility of a full LCA that quantifies multiple types of environmental burdens associated with a given product will automatically follow. In each of the following applications, LCA provides valuable insights for, but does not supplant, benefit-cost analysis.

2.1. Pollution Leakage

The simplest policy rationale for product LCA is pollution leakage. Leakage is simply an increase in pollution outside the jurisdiction of a policy. Typically, it is an unintended negative consequence. The targeted response to leakage is to harmonize environmental regulations across all relevant sources of pollution. In the extreme case of globally well-mixed pollutants, this requires binding environmental agreements between sovereign nations. Given uniform regulations, any unintended increase in pollution from some activities or regions represents merely a pecuniary externality. In practice, such harmonization is either rare, imperfect, or both. A list compiled by the United Nations contains 36 treaties and agreements that are close to achieving universal participation. Among these, the Montreal Protocol on Substances that Deplete the Ozone Layer (**http://ozone.unep.org/en/treaties-and-decisions/montreal-protocol-substances-deplete-ozone-layer**) is the only one that establishes a time line for the global phase-out of chemicals causing a certain type of pollution. This specifically addresses the production of ozone-depleting chemicals, and most recently, with the Kigali Amendment, the phase down in the use of hydrofluorocarbons. There is no other example of a global environmental agreement that obviates concerns about leakage, and with it, any need to focus on product life cycle emissions. The Paris Agreement of 2015 sets out a global action plan for reducing GHG emissions but stops short of specifying binding national emissions targets (**http://unfccc.int/paris_agreement/items/9485.php**). There exist several justifications for subglobal measures to fill this void, which inevitably create concerns about leakage (Rajagopal 2016b).

The consequence of leakage is that importing regions enjoy a cleaner environment while exporting regions experience increasing pollution (Levinson 2009). However, for global externalities, trade-induced leakage could under certain conditions leave even importing regions worse off from an environmental standpoint. First, this could happen when, for example, production of the use of dirtier inputs or less-efficient capital is more pollution-intensive abroad than at home. Calculating the domestic and global changes in pollution, regardless of whether they are due to a change in environmental, labor, or trade policy, requires a life cycle approach. Industry-specific life cycle metrics are also essential to policies for mitigating leakage. Border adjustment policies, such as tariffs on emissions embodied in imports, require estimates of life cycle emissions for imported goods (Fischer & Fox 2012, McAusland & Najjar 2015). Second, when a region happens to be a relatively large consumer but a relatively small producer of a polluting good or activity, then policy makers there could reduce global pollution by raising the cost of consumption in the hope of driving down global demand for goods with high embodied pollution (Bushnell & Mansur 2011). An example of such a regulation is the California LCFS, which sets an upper boundary on the state-average life cycle carbon intensity of transportation fuels consumed within the state (Lemoine 2016). This is a form of vertical targeting where upstream pollution is reduced by targeting downstream activities, which necessitates LCA-like calculations.

2.2. Transaction Costs of Environmental Regulations

By providing an aggregate measure of pollution over space and time, LCA can help reduce the transaction cost of environmental regulation. Here, transaction cost refers narrowly to only the costs of administration, monitoring, and enforcement of a policy.[2] When there is a high transaction cost, achieving the neoclassical ideal, wherein the marginal social cost of pollution equals the

[2]Krutilla (2010) takes a more comprehensive view of the transaction cost of environmental regulation. It includes the ex ante costs of establishing environmental rights, such as the political costs associated with determining the minimum level of environmental quality, whether an emitter has the right to pollute, or whether the victims of pollution have a right to a clean environment, and the costs of choosing the specific policy instruments (e.g., tax, tradable permits, standards). It also includes the ex post costs of administration, monitoring, and enforcement of the adopted regulation. We focus only on the latter to derive a justification for LCA.

marginal social benefit of pollution, can become impractical. One reason for high transaction cost is heterogeneity. When there are many polluters, it is costly to collect data and monitor all sources of pollution (Helfand et al. 2003). This becomes even costlier when the damage from pollution varies with the location and timing of emissions. In such cases, a type of vertical targeting in which a handful of point sources (e.g., mining industries or electric utilities) are held accountable for all the emissions occurring upstream or downstream of their own actions can minimize transaction costs that would be high under textbook solutions. This approach is similar in spirit to LCA except that, instead of estimating life cycle emissions of a finished final good, the focus is on the emissions embodied by a raw material or a primary input. Of course, for this approach to work, emissions downstream of extraction cannot involve too much variability. In other words, for any given type of emissions, total downstream emissions should be a constant proportion of the quantity of the pollutant or its precursor contained in the raw material stage. In the case of fossil fuels, for instance, Metcalf & Weisbach (2009) argue that, by regulating a small number of producers of the primary fossil fuels (e.g., coal, oil, and natural gas) based on the life cycle carbon intensity attributable to their products, more than 80% of domestic GHG emissions in the United States can be regulated. Even with this upstream approach, calculating the optimal upstream tax on polluters located within the home region will require an LCA-like calculation for border adjustments on imported commodities when environmental regulation is not uniform across different political jurisdictions.

A dynamic view of transaction costs can explain policies that make producers of durable goods responsible for management at the end of their product's useful life. Basically, these policies aim to both reduce the externality from improper disposal of toxic materials and slow the rate of depletion of scarce natural resources through recycling, reuse, and redesign. In theory, a combination of product tax, recycling subsidy, and waste disposal fee can achieve the optimal amount of waste disposal, recycled and virgin material use in production, and product design improvements. However, various combinations of policy instruments are used, which can collectively be termed as extended producer responsibility (EPR) (Walls 2006). All such policies rely to a lesser or greater extent on life cycle information specific to a product or the primary materials comprising the product. At a minimum, transaction costs are a motivation for a life cycle approach to pollution reduction, and in some cases, they justify the explicit use of life cycle performance measures to regulate pollution.

2.3. Technology Innovation and Adoption

LCA-based regulations provide incentives to both developers of an innovation and potential adopters (Zilberman et al. 2012). For instance, California's LCFS regulation mandates a reduction in the life cycle GHG intensity of fuels used for transportation within the state. It thus creates incentives for innovation in biofuels, electric vehicles, natural gas vehicles, and fuel cell vehicles, to name a few alternatives to conventional petroleum-derived liquid fuels. Furthermore, there exist multiple technical pathways to producing and utilizing each alternative. Biofuels are a case in point. The combustion of biofuels releases carbon that is recaptured by crops during photosynthesis. However, the production of biofuels (currently) entails fossil fuel use for various activities in the supply chain. Therefore, a decision to support biofuel use in light-duty vehicles, for example, ought to depend on the life cycle GHG intensity of a biofuel relative to that of gasoline, the reference fuel. However, there exist a number of different biofuel pathways that result in different GHG intensities, and therefore, different levels of GHG benefits relative to gasoline (Kendall & Yuan 2013, Spatari et al. 2010). Furthermore, the life cycle GHG intensity of gasoline itself varies depending on the type and location of extraction of crude oil and the technical aspects of the oil refining process (Bergerson et al. 2012, Jaramillo et al. 2007). Likewise, for durable goods, material choices (e.g., whether virgin or recycled materials), the product attributes (e.g., multi-functionality and expected life), and end-of-life options (e.g., landfilled, recycled, or reused) will all lead to different environmental footprints. The CalRecycle Used Oil LCA for the California Oil Recycling Enhancement Act is a specific example in which LCA is being used to guide the adoption of the environmentally least harmful management strategy for used motor lubricants (Reed 2012). This regulation is partially inspired by an LCA that showed that combusting the used oil for power generation might cause 100 times more environmental harm (primarily through heavy metal emissions) relative to rerefining into lubricants or distillation into ship fuels (Boughton & Horvath 2004). In all such applications, LCA provides critical information for public policy to target innovation and adopt the most desirable technology pathways.

2.4. Environmental Information Provision Policies

The ecological attributes of a product, such as the pollution resulting at different stages of its production and at the end of its useful life or the producer's environmental management practices, are credence attributes; i.e., they cannot be verified by consumers either before or after purchase even with repeat purchases (Caswell & Mojduszka 1996). When consumers' information about such attributes is inaccurate, then they suffer disutility. When they are given asymmetric information on ecological preferences, adverse selection might result. LCA can help rectify this asymmetry, which is typically accomplished through various means, including mandatory environmental reporting and disclosure rules, the right to information regulations, and environmental labeling and certification schemes (both private and public). These approaches empower consumers, local communities, investors, and other stakeholders to pressure polluters to change their behavior even in the absence of more conventional types of regulation (Karl & Orwat 1999). The specific approach that utilizes LCA is environmental labeling, or simply, ecolabeling. As far back as 1999, there existed at least 18 ecolabeling schemes worldwide that used LCA (Karl & Orwat 1999), and several more have emerged since.

3. DIFFERENT FORMS OF USE OF LIFE CYCLE INFORMATION

Information generated through LCA could be used in conjunction with any of the standard policy instruments. We classify policy applications of LCA into three broad classes: pure information provision, passive environmental regulation, and active environmental regulation. We discuss each of these below.

3.1. Pure Information Provision

In this type of role, LCA is used to simply apprise decision makers of the life cycle footprint of a product, activity, or an organization. For instance, public agencies could simply provide life cycle information with a view to encouraging voluntary actions by individuals and businesses to reduce their footprint. There is a long history of US government interest in LCA dating as far back as 1974, when it was referred to as resource and environmental profile analysis (Reed 2012). Prior to the adoption of the first mandatory life cycle emissions intensity limit for liquid biofuels as part of the Energy Independence and Security Act 2007, this interest was concentrated in capacity building through the development of tools and guidelines for LCA and their use for information generation (**https://www.epa.gov/laws-regulations/summary-energy-independence-and-security-act**).

A specific example is the US EPA's Sustainable Materials Management program (**https://www.epa.gov/smm/sustainable-materials-management-basics**), which aims to encourage life cycle thinking in business decision making and to develop analytical tools. Government managed or supported ecolabeling schemes that use LCA also fall under this category.

A slightly more coercive use of LCA is to adopt mandatory disclosure rules requiring firms to disclose life cycle information at either the specific-product level or an organizational level. For instance, the rules could require disclosure of one or more types of life cycle pollution, disclosure of the types of materials and processes employed in the production, or disclosure of the location of suppliers. However, firms are not required to take any corrective measures under this type of use. A specific example is the Accounts Act enacted by the Norwegian Parliament in July 1998, that requires firms to disclose how their business affects the environment; the information must include life cycle details of a firm's product or service (Nyquist 2003). The types of information to be disclosed include the energy and raw materials consumed, emissions during transportation, and the effects on the environment during the use phase and at end-of-life.

3.2. Passive Regulation

In this type of role, LCA is used by the government to select specific production practices (e.g., organic production, no-till farming, cage-free management of livestock), specific types of products (e.g., based on recycled content, renewable content, or locally sourced content), or specific end-of-life management practices (e.g., recycling or EPR) for public support. The nature of public support could be as wide ranging as mere recommendations, certification of environmental performance through a labeling or qualification scheme, or a more serious involvement in technology adoption. Examples of the latter include the adoption of mandatory targets or providing financial incentives for production or consumption of a specific type of product (e.g., renewable electricity, electric vehicles) or activities (e.g., specific land management practices such as no-till farming). However, in this type of role, once a specific policy approach is chosen, there is no further use of LCA until perhaps the time when a policy is to be readopted or redesigned. Specific examples of such applications include policies for managing solid wastes, such as EPR (first adopted in Sweden), "take-back" of packaging (first adopted in Germany), and deposit-refund schemes for recycling consumer products, such as beverage containers, consumer electronic goods, automotive batteries, tires, and several other hazardous materials. Last but not least, the California Oil Recycling Enhancement Act aims to support the adoption of most benign waste oil management practices (**http://www.calrecycle.ca.gov/usedoil/policylaw/default.htm**). In these examples, there is little direct use of quantitative estimates of the life cycle environmental performance of one alternative versus another. Contingent on the availability of life cycle information specific to different firms within an industry and the transaction cost associated with verification of this information, the cost-effectiveness of each of the aforementioned types of interventions could be improved.

3.3. Active Regulation

In this type of role, life cycle emissions serve as the basis for an incentive (tax or subsidy), tradable permits, or an environmental performance standard. This necessitates the use of LCA as a tool to verify the level of emissions on which the facility should be taxed, for which it is eligible to receive a subsidy, or to determine whether the entity complied with a given life cycle emissions performance standard. One set of policies that involves such a use of life cycle information is fuel emissions intensity regulations, which started with the United Kingdom's Renewable Transport Fuel Obligation (**https://www.gov.uk/guidance/renewable-transport-fuels-obligation**). These have since also been adopted by the states of California (i.e., the LCFS regulation) and Oregon in the United States (**https://www.oregon.gov/deq/aq/programs/Pages/Clean-Fuels.aspx**) and the Canadian province of British Columbia (**http://www2.gov.bc.ca/gov/content/industry/electricity-alternative-energy/transportation-energies/renewable-low-carbon-fuels**). The RFS is another policy that actively regulated life cycle GHG emissions from biofuels. Outright bans based on life cycle performance also directly use LCA. One example is the European Union's Restriction of Hazardous Substances Directive, which bans the use of lead, mercury, brominated flame retardants, and several other hazardous substances that could find use in electrical and electronic equipment (**https://ec.europa.eu/growth/single-market/european-standards/harmonised-standards/restriction-of-hazardous-substances_en**). The US RFS, which sets targets for consumption of different classes of biofuels, precludes the use of biofuels exceeding a certain threshold level of life cycle GHG intensity as a compliance option (**https://www.epa.gov/renewable-fuel-standard-program**). When enacted, border adjustment policies that are based on embodied pollution in imports or the pollution avoided by cleaner domestic goods will also be examples of the active regulatory use of LCA.

The three different classes of LCA applications require increasing levels of accuracy of life cycle information. Financial incentives (taxes, subsidies, and tradable permits) and performance standards based on LCA require verifiability of claims, which require an ability to accurately determine the life cycle emissions of a specific facility, firm, or product. But the controversies surrounding the GHG emissions intensity of biofuels show how difficult this can be. Therefore, whereas transaction costs of implementing neoclassical solutions provide a justification for life cycle–based approaches, certain forms of uses entail nonzero transaction costs, specifically those that fall within what we call active regulation. Therefore, decision makers need to be clear about the rationale for using an LCA-based approach and also consider costs and benefits of alternative types of uses of life cycle information.

The discussion thus far presents the economic case for LCA from a theoretical perspective. The remainder of the review is devoted to the practical issues that arise when conducting an LCA and directions for related future research.

4. ISSUES IN THE USE OF LIFE CYCLE ANALYSIS ESTIMATES IN ENVIRONMENTAL REGULATION

Although LCA methods and applications have become more refined over the last few decades, there remain both methodological (Section 4.1) and implementation-related (Section 4.2) challenges when it comes to using the numerical estimates for environmental regulation. The methodological challenges pertain to the definition of the true life cycle footprint of a product, whereas the operational and implementation-related challenges span the entire policy life cycle. Addressing these challenges is critical for the effective use of LCA, whether it be for information provision or for passive and active regulatory use. McManus et al. (2015) provide a valuable overview, targeted at an LCA audience, of challenges identified to be at the intersection of biofuels/bioenergy LCA and policy, although many of the challenges transcend the bioenergy area and are applicable to LCA and policy more generally. This section discusses, from our perspective, the key challenges that LCA faces in gaining widespread acceptance and use by economists, policy analysts, and decision makers and then provides recommendations for overcoming them. However, these challenges should not be interpreted as a deterrent to any form of using LCA in policy.

4.1. Methodological Challenges

4.1.1. Modeling emissions due to price effects. Traditionally, LCAs focused on the direct and indirect burdens that are traceable to the activities and facilities required for manufacturing, delivering, consuming, and managing a product's end-of-life at the level of a single household, firm, or the average (or marginal) output of an industry. However, a different channel of indirect emissions[3] arises from the effect of expansion of a new industry: prices throughout the economy. For instance, appropriating either crops or cropland for biofuels will increase food prices (Chakravorty et al. 2009). This will increase the supply of cropland at the expense of idle land, grazing land, or forest land, which is the so-called indirect land-use change (ILUC) effect (Hertel et al. 2010, Khanna & Crago 2012, Searchinger et al. 2008). Likewise, the effect of biofuels on fuel prices is the source of the so-called indirect fuel use emissions (Bento et al. 2015, de Gorter & Drabik 2011, Rajagopal 2013, Rajagopal et al. 2011). As another example, Sedjo & Swallow (2002) show theoretically why eco-certification programs for forestry products might improve the ecological state of isolated forested areas while degrading the overall ecosystem within which eco-certified goods are produced. From an economic perspective, these are simply general equilibrium effects.

In the LCA literature, inclusion of emissions due to price effects distinguishes attributional LCA (ALCA) from consequential LCA (CLCA). ALCA is the original type of LCA, which focuses on estimating the emissions associated with each stage of the life cycle (from "cradle to grave") of a product or service at a given point in time. ALCA studies may examine a single product with the aim to reduce emissions associated with the product life cycle and to avoid burden shifting (reducing emissions of Pollutant X from one life cycle stage but increasing emissions of Pollutant X or Pollutant Y from another stage) by capturing both direct and indirect impacts and considering all life cycle stages and a range of impacts. ALCA can also be used to compare life cycle emissions of products that provide the same function or close to the same function. In all types of LCA, a specified functional unit provides a reference to which the inputs and outputs are related. When different systems are assessed, the functional unit aims to ensure comparability, but in practice, attributes often differ among products being compared, even if a common functional unit is utilized. For example, gasoline and battery electric vehicles are often compared on the basis of the functional unit of one vehicle kilometer driven. However, gasoline and battery electric vehicles generally have different attributes (e.g., driving range, noise level); thus, a consumer driving one kilometer in one vehicle versus the other may not view these vehicles as equivalent or providing the same service. Attention to the equivalency of products being compared is a notable issue in LCA. Overall, there is a large history of ALCA studies, some dating back decades, and others being conducted currently (see Matthews et al. 2015 for further detail on the history of ALCA). A few recent examples of ALCAs comparing different systems include light-duty vehicles utilizing lignocellulosic ethanol versus those using biobased electricity (Luk et al. 2013); conventional hand dryer versus paper towel as hand-drying methods (Joseph et al. 2015); and multilayer cartons versus polyethylene bottles for packaging milk (Bertolini et al. 2016).

In contrast to ALCA, CLCA has attracted attention in recent years and assesses the changes in total emissions as a consequence of a decision (e.g., change in demand for a product, change in policy). Rajagopal (2014) conceptually distinguishes CLCA from ALCA by explaining that ALCA focuses on vertical dependencies in a product's supply chain. In contrast, CLCA accounts for the horizontal linkages (e.g., competition for a good in alternative applications) at each step in the vertical supply chain, and hence, is broader in scope.

[3]Refer to the introductory paragraph for a traditional interpretation of indirect emissions in the LCA context.

Carrying out a CLCA is challenging for the following reason. Price effects arise from the interaction of market supply and market demand for any given commodity or service. When the market for a given commodity (or the intermediate inputs to its production) is global, then estimating the ceteris paribus effect of regional or national policy (e.g., RFS) on global prices and emissions in various commodity markets (e.g., land, different types of crops and fuels) necessitates the use of multimarket partial equilibrium (PE) or computable general equilibrium (CGE) modeling tools. It is well known that the estimates derived from such tools are sensitive to the model inputs and assumptions (including functional forms, model parameters, and calibration exercise). As the number of commodities affected by a policy increases, the number of such assumptions increases, and typically, there is little empirical basis to justify all such assumptions. As a result, estimates from CLCA can be wide ranging and lead to a lack of consensus.

A challenge associated with LCA methods is the ongoing debate regarding the appropriate uses of ALCA and CLCA in policy applications, with much of the debate focused on their use with regard to climate change mitigation and biofuels policies (Anex & Lifset 2014; Brandao et al. 2014; Dale & Kim 2014; Hertwich 2014; Plevin et al. 2014a,b; Suh & Yang 2014). ALCA and CLCA have different goals and therefore aim to provide different information. Although some feel that ALCA is not relevant for use in policy because it does not focus on changes associated with a particular decision (e.g., implementation of a policy) and does not include market-mediated effects, in our view, ALCA and CLCA are complementary and have relevant roles in policies that incorporate LCA (Rajagopal 2014). ALCA yields estimates of life cycle emissions for a single polluter, which can be utilized to determine whether the entity is in compliance with a specific standard's target, such as in the case of the EU Fuel Quality Directive (FQD). As a complementary approach, insights from CLCA will advise policy makers about the vulnerable aspects of a policy with regard to environmental impact or consequences of decisions and assist them to modify potentially counterproductive formulations early in the policy life cycle (for additional details, see Rajagopal 2014). Essentially, the appropriate type(s) of LCA and how to best use their results in LCA are key research questions. CLCAs present some unique challenges not only in their implementation but also in operationalizing their use in regulation, a point we return to in Section 4.2.

4.1.2. System boundary selection. The selection of an appropriate system boundary for any LCA, i.e., determining which activities are included in the analysis, is a challenging but essential task to ensure that the study accomplishes its stated goal. System boundaries have attracted much attention as a result of their often arbitrary selection due to resource and data constraints (Hendrickson et al. 2006) as well as because they and methods for their selection differ between ALCA and CLCA. Accurately defining the system boundary is essential for producing reliable results and important for LCA's effective use in policy, irrespective of whether the LCA is used for information provision or passive or active regulation. Therefore, it can be argued that consistent and appropriate/justifiable system boundary selection becomes increasingly important as one moves from information provision to active regulation. The latter category is the most critical, as the quantitative LCA results will be used in determining a regulated party's compliance, such as in the LCFS.

The specification of system boundaries falls within the first phase (goal and scope definition) of the ISO's 14040 Environmental Management–Life Cycle Assessment (ISO 2006a). Although the second part of the Assessment (ISO 2006b) presents guidance to delimit the system boundary, the decisions are essentially left to a practitioner's judgment. In addition, the system boundary definition and guidance under the ISO are applicable to ALCA and not CLCA, and more specifically, a process-based approach to ALCA, rather than Economic Input-Output LCA (EIO-LCA) or hybrid approaches (for further detail on these latter approaches, see below). In ALCA, the system boundary aims to capture physical flows (e.g., emissions) that are directly traceable to the supply chain of a product, its use, and end-of-life phases. In contrast, the system boundary of CLCA extends beyond the supply chain and incorporates aspects of the technical system that will change in response to changes in the product life cycle, such as increased demand for the product.

According to Finnveden et al. (2009, p. 6), "The system boundaries in a consequential LCA are defined to include the activities contributing to the environmental consequences of a change, regardless of whether these are within or outside the cradle-to-grave system of the product investigated." CLCAs often incorporate economic concepts, such as marginal costs and data and market-mediated effects (Ekvall & Weidema 2004), which can result in expansion of the system boundary, as the studies generally estimate impacts over wider geographical and temporal ranges than ALCA. Because it is infeasible to model the entire global economy in detail, CLCA studies often utilize PE or CGE models (Earles & Halog 2011, Khanna & Crago 2012, Kløverpris et al. 2008). Whereas CLCAs often have a broader system boundary than ALCAs, in CLCA, only those elements of a system that are affected by the decision(s) are included in the analysis (Tillman 2000). Therefore, the boundary may in some cases be narrower than in an ALCA. To date, there is no single generally used or accepted modeling framework or guidelines for CLCA (Rajagopal 2016a).

In the mid-1990s, prior to the formal development of CLCA, the Green Design Institute at Carnegie Mellon University pioneered a technique called EIO-LCA, which as the name suggests, builds upon national income accounts (Hendrickson et al. 2006). Because an input-output table records intersectoral linkages across the entire economy, EIO-LCA captures the direct effects as well as all indirect effects across the entire supply chain, barring effects that might arise from the effect on commodity prices. This is because of the Leontief structure imposed by this approach. The approach is used for analyzing the effect of small shocks and over relatively short time horizons when the structure of the economy can be assumed to be fixed. The EIO-LCA approach eliminates the arbitrariness in a system boundary definition that is associated with any other approach to LCA. Hybrid approaches, which combine aspects of EIO and process-based LCA, can help strike a balance between technological richness and truncation errors associated with system boundary definition. However, EIO-LCA and hybrid approaches are in principle ALCA and not CLCA for the reasons noted above.

Most of the focus on system boundaries in LCA has been on physical boundaries rather than legal boundaries, the latter of which are key for policy. Vandenbergh & Cohen (2010) note that leakage occurs when there is a mismatch between physical and legal boundaries. They use the example of carbon reporting standards to illustrate the critical nature of this issue, noting that the majority of standards have not required the reporting of supply chain emissions; the potential implication of this omission is substantial leakage.

Regulators must clearly specify the system boundary for life cycle emissions and justify this selection, keeping in mind that the boundary definition might determine whether a product is more or less pollution intensive relative to another. Though a wider boundary will capture more of the system impacts, it increases the complexity of the analysis and uncertainty (Ney & Schnoor 2002).

4.1.3. Joint production. Differences in the treatment of jointly produced products in an LCA is an oft-cited reason for divergence of LCA results for the same product from one study to another (Gerbrandt et al. 2016), and studies comparing different methods of treating jointly produced products have found large differences in results (Wang et al. 2011). Treatment of jointly produced products can be the main driver behind a product's LCA performance. Consequently, how joint production is treated in LCA is a key issue for its effective use in policies. Joint production is often referred to as coproduct treatment in the LCA literature. The main issue is deciding how the environmental burdens of the activities within the system boundary should be assigned when there is joint production. The ISO (2006b) provides guidelines regarding the treatment of jointly produced products and generally recommends that allocation be avoided wherever possible by dividing multifunctional unit processes into subprocesses or through system expansion. In reality, most multifunction systems (e.g., food processing facilities, oil refineries) include processes that are common for some or all of their functional outputs, and therefore, this method rarely avoids allocation but instead reduces the allocation problem (Furuholt 1995, Høgaas Eide & Ohlsson 1998). The ISO (2006b, p. 14) states that,

> Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the underlying physical relationships between them; i.e., they should reflect the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system.

Finally, if that is not possible, the standard suggests partitioning with other relationships (e.g., economic value). However, in practice, this hierarchy is not always easy to implement. For additional detail on joint production methods, see Matthews et al. (2015). In the remainder of this section, we focus on aspects most relevant to the use of LCA in policies.

System expansion is the recommended procedure for CLCA, although not all LCAs that incorporate this approach would be classified as CLCAs. System expansion involves expanding the product system to include alternative method(s) of producing the jointly produced product(s) (or functions of the product). For example, if a lignocellulosic ethanol facility produces both ethanol and electricity, then rather than allocating emissions associated with the production of these products to each of them based on, for example, their energy content, system expansion can be utilized. With this approach, all of the emissions will be attributed to ethanol, the primary product, but an emissions credit (negative value) will be given to the ethanol for the facility's production of electricity if it is exported to the grid. This credit will be based on the quantity of emissions that would have resulted if the electricity had to be produced using the grid. In practice, system expansion is possible for a wide range of situations, but it has several challenges. First, it can be difficult to identify a realistic substitution scenario due to the uniqueness of the production process (Niederl-Schmidinger & Narodoslawsky 2008). Second, it also has considerable data requirements because the alternative product/production method must be included in the LCA boundary, and the number of what-if assumptions can be large enough to lead to divergent LCA results (Heijungs & Guinée 2007).

Joint production issues come into play in product-based regulations in several ways. There are various examples from low-carbon fuel regulations. Feedstocks for biofuel production (e.g., corn and corn stover) can be jointly produced, the latter comprising the stalks and leaves after harvest of the grain. Currently, stover is viewed as a by-product of corn production and has typically been left on the field, not having a significant market. Most LCAs of corn stover ethanol production have not allocated emissions associated with corn production to the stover; instead, all emissions associated with corn production have been allocated to the grain (Gerbrandt et al. 2016). If emissions are instead allocated to both grain and stover, this influences LCA results for the stover ethanol, as corn production is an energy-intensive process. As noted above, there is often joint production at the biofuel production facility. For example, the majority of corn ethanol is produced jointly with Distillers' Dried Grains and Solubles (DDGS), which is an animal feed. Many LCAs have used system expansion (or displacement) to provide a credit to the corn ethanol related to the DDGS assumed to displace soybean meal. The magnitude of these credits can be large, thus considerably effecting LCA results.

Different regulations and different jurisdictions have adopted distinct methods of treating joint production. For example, the California LCFS and federal RFS focus on displacement (similar to system expansion) (Wardenaar et al. 2012), whereas the EU FQD specifies allocation on an energy basis as the most appropriate (**http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32009L0030**). The FQD recommends that results for the substitution approach also be reported but states that these are generally comparable with those produced by the energy allocation method. Prior literature, however, has not always supported this view that allocation and system expansion/substitution give similar results (e.g., McManus et al. 2015). For policy applications, the consistency, transparency, and justification for the selection of methods for treating joint production are ongoing issues, and decisions in this realm can have nontrivial implications for products aiming to meet emissions standards, particularly in the active regulatory category of application.

4.1.4. Variability and uncertainty. All aspects of LCA (e.g., data, models, and scenarios) have associated variability and/or uncertainty (Steinmann et al. 2014). Variability can arise due to heterogeneity (e.g., quality of inputs, vintage of capital) and spatial and temporal differences, whereas uncertainty arises due to randomness and fundamental gaps in our knowledge. Explicit characterization of uncertainty in LCA is required to support reliable results and informed decision making; however, it has often been left out of studies or incompletely examined (Lloyd & Ries 2007, Williams et al. 2009). Policies falling into the active regulatory category have generally lacked explicit characterization of uncertainty in LCA. Those in the information provision and passive categories have relied on studies that generally do not provide detailed examination of uncertainty, although some of these have included scenario/sensitivity analysis (e.g., CalRecycle Used Oil LCA for the California Oil Recycling Enhancement Act). The RFS and the California LCFS, which fall into the active category, both compared life cycle GHG emissions of incumbent and low-carbon fuels based on point estimates and did not include uncertainty in the estimates. However, variability and uncertainty assessment revealed a 10–35% probability that the life cycle GHG emissions of natural gas vehicles (cars and buses) are higher than those of vehicles powered by gasoline or diesel (Venkatesh et al. 2011). Additionally, Mullins et al. (2011) found large ranges for life cycle GHG emissions associated with ethanol and butanol, and each fuel showed some probability of resulting in greater emissions than gasoline. These analyses suggest that, without explicit incorporation of uncertainty in LCA in active regulations, fuels may be classified as low carbon, when in reality, they are not. Additionally, studies have reported the superiority on a life cycle basis of one product over another, but they do so without an examination of uncertainty.

The ISO (2006a) recommends sensitivity and uncertainty analyses for studies intended to be disclosed to the public, which is obviously applicable to all policy applications, but it provides no specific guidance in this regard. Methods typically used to examine uncertainty include sensitivity analysis, scenario analysis, and Monte Carlo simulation. Although explicit quantification of uncertainty has its benefits, there are trade-offs with added complexity and implementation/communication challenges when results are no longer point estimates. These issues are discussed in the next section.

4.2. Implementation-Related Challenges

Equally important as overcoming methodological challenges associated with LCA is addressing the challenges of using LCA information in public policies. These challenges can occur throughout the policy cycle, beginning with the justification for an LCA-based regulation through verifying compliance with a regulation, and ultimately, an ex post impact assessment. In light of the recent and most challenging policy uses of LCA, the focus here is on the implementation-related challenges in using life cycle results to design or implement an emissions performance standard or market-based policies, such as tax, subsidy, or tradable permit systems.

4.2.1. Lack of confidence in life cycle analysis as a tool and its consequences. As with any modeling-based exercise, there is skepticism that one can obtain essentially any desired outcome from an LCA (recall debates over whether corn ethanol reduces GHG emissions compared to gasoline; e.g., see Farrell et al. 2006). Although LCA data, methods, and study quality have improved, and critical reviews have become more common as the field matured, challenges remain in the implementation of ALCA and CLCA. LCAs have often been criticized for not being robust and/or for lacking transparency. The latter occurs due to the nature of the method and the lack of care of some analysts in documentation. It is critical that those involved in the policy process have confidence in LCA as a tool and in the specific results of studies that will be used in the process.

Ensuring that an LCA utilized in a policy process is performed in an iterative manner is essential, but the iterative nature of LCA is often overlooked. Sensitivity or uncertainty analysis may motivate an LCA practitioner to go back to collect more data, refine an analysis, or repeat the analysis until results can support the original goal; if data are too incomplete, then conclusions or recommendations may not be warranted (Curran 2013). The iterative process also extends to interactions with stakeholders (see Section 4.2.4), where feedback on the LCA is provided throughout the process, and iteration is undertaken as needed.

The ISO (2006a) requires that a critical review is performed of any LCA used to make a comparative assertion that is disclosed to the public. The review is intended to verify specific aspects, including that the methods used are in compliance with ISO guidelines and are scientifically and technically valid. In addition, it requires that data are appropriate, interpretation is accurate, and the report is transparent and consistent. The review aims to enhance scientific quality and increase the transparency of the LCA, thereby improving credibility of the study for stakeholders (e.g., CalRecycle Used Oil LCA for the California Oil Recycling Enhancement Act). Although a critical review as well as a more broad compliance with ISO guidelines in completing the LCA may improve confidence in an LCA in some cases. In others, it could be viewed as more of a procedural exercise and one that does not really identify the essence of the issue; i.e., all procedures may be followed in doing the LCA but the outcome is biased by any one of a number of issues.

4.2.2. Mitigating emissions due to price effects. Emissions due to price effects pose special challenges in that, although they may be real (i.e., not simply pecuniary) externalities, they are challenging to regulate. Recall the discussion in Section 4.1.1 with respect to modeling- and estimation-related issues. The operational challenge is that, because it is hard to verify empirically any estimate of emissions due to price effects, affixing the entire responsibility for them on a small number of regulated entities can appear excessively onerous (Zilberman et al. 2011). It could lead to the affected parties litigating the estimates adopted by regulators, thereby increasing the transaction cost of regulating such emissions. Both excessively high and excessively low estimates can affect investment in innovation. Policy makers need to carefully weigh the benefits and costs of alternative approaches to dealing with indirect emissions without ignoring them altogether because they are uncertain. One such alternative is to adopt more stringent standards for life cycle emissions in the traditional sense of what is traceable to one's own supply chain while avoiding explicit regulation of emissions due to price effects. Another alternative could be to adjust downward the volumetric targets for technologies or products that appear to entail a high risk of indirect emissions. With this approach, policy makers could buy themselves additional time to undertake further research, and at the same time, allow emerging industries to gain from learning-by-doing and network effects, which can counteract harmful unintended effects.

4.2.3. An inadequate choice of alternatives. Another issue in implementing LCA in policy results from the LCA not evaluating alternatives that some stakeholders feel should be included, leading to disagreement among the stakeholders. Because the stakeholders in a policy process come from diverse organizations (e.g., government, industry, consumer interest groups), they have different interests and perceptions of the situation. Bras-Klapwijk (2003) illustrates examples in which important alternatives were not included in LCAs, thus leading to nonacceptance of the LCA conclusions by some stakeholders. There are no official guidelines or tools in LCA on the selection of alternatives. However, Bras-Klapwijk developed some recommendations, including attention to determining the functional unit, as this assists in deciding which products can be reasonably compared; participation of the full range of stakeholders in the entire LCA process to obtain their insights and generate alternatives; formulation of the problem to also include nonenvironmental aspects, which would lead to alternatives that are attractive from others' points of view; and assurance that screening and selection of alternatives are done by applying relevant criteria. We would also add to this list the assurance that LCA is carried out as an iterative process (as discussed in Section 4.2.1) and facilitates inclusion of relevant alternatives.

4.2.4. Communication and engagement of practitioners, policy makers, and other stakeholders throughout the life cycle analysis process. Clear communication and engagement among LCA practitioners, policy makers, and other stakeholders in the policy process are needed throughout the LCA to ensure its effective use in policy. Seidel (2016) contrasted the use of LCA in two public policy case studies, the CalRecycle Used Oil LCA for the California Oil Recycling Enhancement Act (see Sections 2.3, 3, 4.1.4, and 4.2.1) and the Alberta scrap tire recycling example in Canada. They found that the high level of stakeholder involvement from the beginning of the study in the California case resulted in a much better understanding and trust from all parties involved and an increased chance of incorporating LCA in public policy.

4.2.4.1. Life cycle analysis concepts and methods. Specific terminology and methods used in LCA (e.g., system boundary, system expansion, functional unit, weighting, life cycle impact assessment, ALCA, CLCA, and ILUC) may be confusing to policy makers and stakeholders not familiar with LCA (Reed 2012, Seidel 2016). Additionally, they may not always be aware of the complexity of LCA. For effective use in policy, there is a need to introduce these aspects to stakeholders early in the process. LCA is sometimes confused with life cycle costing, risk assessment, and benefit-cost analysis. The concept of LCA overall, as well as its focus on environmental aspects and its lack of monetary bases, should be explained to stakeholders. This would ensure clarity in their expectations of the types of insights the LCA can provide. The results of an LCA represent just one of many inputs into a decision-making process and must be utilized with outcomes of other analysis tools and information (Curran 2013). Understanding the role that LCA can play in the policy process, including its strengths and weaknesses, is critical for its effective use.

4.2.4.2. Life cycle analysis study results. A major challenge facing LCA experts is communicating study results in a clear and effective way to nonspecialists (Cowell et al. 2002, Reed 2012). LCA often comprises complex frameworks, and in most cases, a multitude of different data sources and assumptions as well, which can be difficult for nonspecialists to interpret. In addition, one product could be preferred over another with respect to one impact assessment category but could have the opposite result for another (e.g., Product A has lower global warming potential but higher acidification potential than Product B). Although there are methods to calculate an overall (single-value) score for impact, they generally involve a loss of transparency in the result and considerable value judgment. LCA experts have to summarize large amounts of data without losing important information, explain multiple scenarios, assumptions, uncertainty, and trade-offs, as well as make the format of the results understandable to diverse stakeholders. There is no commonly accepted practice on how to present LCA results or develop a communication strategy. Nevertheless, some efforts have been made in this regard. The Society of Environmental Toxicology and Chemistry, an organization involved in the development of initial guidelines for LCA (Consoli et al. 1993), more recently identified existing results visualization techniques (e.g., network diagrams, dynamic visualization) as being relevant for presenting LCA results (Laurin et al. 2016). Regarding the communication of uncertainties in LCA, Seidel (2016) recommends presenting them transparently throughout the entire LCA process.

4.2.5. The policy-making process. The incremental and fragmented nature of the policy-making process, combined with Knaggård's (2014, p. 22) observation that fully rational decision making within the process is "no more than ideal," makes it challenging for decision makers to integrate LCA into policies. Cohen (2013, p. 227) stated, "The challenge for the LCA community is to accept the reality of incremental policy-making while crafting new environmental information that fits into the incremental pattern by insulating itself from the perils of partisan politics." A more active collaboration between policy makers and LCA experts throughout the entire policy process may facilitate inclusion of LCA results in a policy.

Although an understanding of uncertainty in LCA is critical, it could be a barrier to the use of LCA in public policy, as policy makers prefer precise and unequivocal scientific information (Cohen 2013, Hammond et al. 1983). Large uncertainty could also lead to a polarization among stakeholders that do not share the same interests. To overcome issues related to uncertainty, there is a need to decrease uncertainty and/or better accommodate uncertainty in the policy process. As our understanding of the natural world improves and methodological aspects are refined, Cohen (2013) suggests that uncertainty would decrease, enabling policy making. However, in some cases, obtaining more knowledge is not feasible or is too resource consuming. In addition, reducing and better accommodating uncertainty have not always improved the policy making process. A study of the Swedish climate change policy process from 1975 to 2007 showed that the main problem in the process was that scientific knowledge was not framed in a politically accessible way (and therefore it was unhelpful as policy advice); it was not an issue of scientific uncertainty (Knaggård 2014).

4.2.6. The cost of life cycle analysis. Last but not least, LCA is generally data and time intensive or simply costly. Furthermore, tolerance for uncertainty in estimates of pollution or pollution intensity diminishes rapidly as its use in regulation changes from pure information provision to active regulatory use, such as when providing financial disincentives (or incentives) or for determining compliance with a performance standard. There are different categories of LCA studies; depending on the goal and scope of the study, the level of detail will be determined so that it corresponds with the study goal and scope. There is no overall consensus in the literature or among LCA practitioners on the categories of LCA studies. The EeBGuide of the European Commission (**http://www.eebguide.eu/**) defines three LCA study categories: screening, simplified, and complete. SimaPro, a popular LCA software, categorizes studies into screening, internal, and those for external communication and continuous use of LCA information (**http://www.presustainability.com/download/SimaPro8IntroductionToLCA.pdf**). The categories of studies typically vary based on the completeness of the assessment, selection of the system boundaries, comparability, data representativeness, and documentation and communication of the LCA results; they also vary in whether adherence to ISO standards and independent review are required. The requisite time and cost of conducting a study vary based on a number of aspects, including those noted above. Based on our industry knowledge, time and costs could range from a few weeks and several thousand dollars for a screening LCA to more than a year and more than US$1 million for a complete LCA for external communication.

Compared to ALCAs, CLCAs require an estimation of market-mediated emissions; they are generally expected to be the most costly and have their outcomes more heavily contested. This is reflected in the $2.5 million, two-and-a-half-year process undertaken in the CalRecycle Used Oil LCA (Seidel 2016). In addition to a comprehensive LCA, the study included a broad stakeholder engagement process, an economic analysis, and an independent review.

Stakeholders need to see the value of spending perhaps $1 million and one year or more to complete a detailed LCA among multiple options to support economic decision making. Stakeholders may need to be convinced of the LCA's value and that it is a cost-effective use of funds. Taking another perspective, the cost of an LCA is likely to be very small in comparison to the cost to society of a policy that has unintended negative consequences that could have been avoided had insights from an LCA informed its development. Overall, it is important to balance the marginal cost of completing an LCA or a more detailed LCA versus the corresponding marginal benefit of the additional information that is provided. This ensures that stakeholders in the policy process understand the utility of LCA through appropriate communication and that LCA is undertaken as an iterative process.

5. CLOSING THOUGHTS

The salience of LCA is its usefulness in identifying and quantifying the unintended environmental consequences of adopting a new innovation. Although LCA differs from conventional approaches to pollution control, let alone textbook environmental policy instruments, the rationale for its use in conjunction with other standard tools and procedures is clear. At least four broad situations—pollution leakage, transaction costs of environmental regulations, support of innovation and technology adoption, and asymmetric information about environmental attributes—warrant attention to the life cycle environmental footprint of specific goods and services.

The information generated through an LCA could find different uses in environmental regulation. The simplest application of LCA is as a tool for deriving qualitative insights or order of magnitude estimates of the change in pollution that might result from adopting a new innovation. The more challenging uses of LCA involve its use to establish performance standards or in the future to tax, subsidize, or allocate permits to pollute based on an estimated life cycle footprint of a product or a firm. The main challenge here is that ex post verification and true up of ex ante estimates is difficult. Such uses of LCA are proving particularly challenging, as shown by the recent experience with policies such as the RFS and LCFS. The application of LCA to other types of products and commodities might entail less complexity relative to its application to biofuels, which have been the focus of many LCAs within the aforementioned policies. Identifying the product types (such as fuels, electricity, and durable consumer goods), policy jurisdiction (whether local, state, national, or a combination thereof), and the type of policy use of LCA that is most appropriate for each combination (whether it is pure information or passive or active regulatory use) is an avenue for future research. A key question in this research concerns the approach, both from estimation and regulation perspectives, for dealing with so-called indirect or market-mediated emissions that arise from the effect of technology adoption on market prices. The literature on the economics of innovation and technology adoption, transaction cost, and benefit-cost analysis contain important insights for this research. This is an area ripe for collaboration between economists, policy researchers, and experts on LCA.

DISCLOSURE STATEMENT

The authors are not aware of any affiliations, memberships, funding, or financial holdings that might be perceived as affecting the objectivity of this review.

ACKNOWLEDGMENTS

D.R. acknowledges the contributions of Professor David Zilberman and late Professor Alex Farrell to his research on the economics of LCA, which originated as a PhD dissertation topic. H.L.M. acknowledges the influential role of the late Professor Lester Lave, Professor of Economics at Carnegie Mellon University, her PhD advisor and mentor, in advancing the fields of economics, LCA, and public policy, as well as in guiding her exploration of these topics. H.L.M. and C.V. thank the Natural Sciences and Engineering Research Council of Canada and BioFuelNet Network Centres of Excellence for financial support.

LITERATURE CITED

Anex R, Lifset R. 2014. Life cycle assessment different models for different purposes. *J. Ind. Ecol.* 18:321–23

Bento AM, Klotz R, Landry JR. 2015. Are there carbon savings from US biofuel policies? The critical importance of accounting for leakage in land and fuel markets. *Energy J.* 36(3). **https://doi.org/10.5547/01956574.36.3.aben**

Bergerson JA, Kofoworola O, Charpentier AD, Sleep S, MacLean HL. 2012. Life cycle greenhouse gas emissions of current oil sands technologies: surface mining and *in situ* applications. *Environ. Sci. Technol.* 46(14):7865–74

Bertolini M, Bottani E, Vignali G, Volpi A. 2016. Comparative life cycle assessment of packaging systems for extended shelf life milk. *Packag. Technol. Sci.* 29(10):525–46

Boughton B, Horvath A. 2004. Environmental assessment of used oil management methods. *Environ. Sci. Technol.* 38(2):353–58

Brandão M, Clift R, Cowie A, Greenhalgh S. 2014. The use of life cycle assessment in the support of robust (climate) policy making: comment on "using attributional life cycle assessment to estimate climate-change mitigation. . ." *J. Ind. Ecol.* 18:461–63

Bras-Klapwijk RM. 2003. Procedures and tools for generating and selecting alternatives in LCA. *Int. J. Life Cycle Assess.* 8(5):266–72

Bushnell JB, Mansur ET. 2011. Vertical targeting and leakage in carbon policy. *Am. Econ. Rev.* 101(3):263–67

Caswell JA, Mojduszka EM. 1996. Using informational labeling to influence the market for quality in food products. *Am. J. Agric. Econ.* 78(5):1248–53

Chakravorty U, Hubert M-H, Nøstbakken L. 2009. Fuel versus food. *Annu. Rev. Resour. Econ.* 1:645–63

Cohen SA. 2013. Life cycle assessment and the U.S. policy-making context. In *Emerging Technologies: Socio-Behavioral Life Cycle Approaches*, ed. N. Savage, M. Gorman, A. Street, pp. 217–44. Boca Raton, FL: Pan Stanford

Consoli F, Allen D, Boustead I, Fava J, Franklin W, et al. 1993. *Guidelines for Life-Cycle Assessment: A Code of Practice*. Pensacola, FL: SETAC

Cowell SJ, Fairman R, Lofstedt RE. 2002. Use of risk assessment and life cycle assessment in decision making: a common policy research agenda. *Risk Anal.* 22(5):879–94

Curran MA. 2013. Life Cycle Assessment: a review of the methodology and its application to sustainability. *Curr. Opin. Chem. Eng.* 2(3):273–77

Dale BE, Kim S. 2014. Can the predictions of consequential life cycle assessment be tested in the real world? comment on "using attributional life cycle assessment to estimate climate-change mitigation..." *J. Ind. Ecol.* 18(3):466–67

de Gorter H, Drabik D. 2011. Components of carbon leakage in the fuel market due to biofuel policies. *Biofuels* 2(2):119–21

Earles J, Halog A. 2011. Consequential life cycle assessment: a review. *Int. J. Life Cycle Assess.* 16(5):445–53

Ekvall T, Weidema BP. 2004. System boundaries and input data in consequential life cycle inventory analysis. *Int. J. Life Cycle Assess.* 9(3):161–71

Farrell AE, Plevin RJ, Turner BT, Jones AD, O'Hare M, Kammen DM. 2006. Ethanol can contribute to energy and environmental goals. *Science* 311(5760):506–8

Finnveden G, Hauschild MZ, Ekvall T, Guinée J, Heijungs R, et al. 2009. Recent developments in life cycle assessment. *J. Environ. Manag.* 91(1):1–21

Fischer C, Fox AK. 2012. Comparing policies to combat emissions leakage: border carbon adjustments versus rebates. *J. Environ. Econ. Manag.* 64(2):199–216

Furuholt E. 1995. Life cycle assessment of gasoline and diesel. *Resour. Conserv. Recycl.* 14(3–4):251–63

Gerbrandt K, Chu PL, Simmonds A, Mullins KA, MacLean HL, et al. 2016. Life cycle assessment of lignocellulosic ethanol: a review of key factors and methods affecting calculated GHG emissions and energy use. *Curr. Opin. Biotechol.* 38:63–70

Hammond KR, Mumpower J, Dennis RL, Fitch S, Crumpacker W. 1983. Fundamental obstacles to the use of scientific information in public policy making. *Technol. Forecast. Soc. Change* 24(4):287–97

Heijungs R, Guinée JB. 2007. Allocation and "what-if" scenarios in life cycle assessment of waste management systems. *Waste Manag.* 27(8):997–1005

Helfand GE, Berck P, Maull T. 2003. The theory of pollution policy. *Handb. Environ. Econ.* 1:249–303

Hellweg S, Milà i Canals L. 2014. Emerging approaches, challenges and opportunities in life cycle assessment. *Science* 344(6188):1109–13

Hendrickson CT, Lave LB, Matthews HS. 2006. *Environmental Life Cycle Assessment of Goods and Services: An Input-Output Approach*. Washington, DC: Resour. Future

Hertel TW, Golub AA, Jones AD, O'Hare M, Plevin RJ, Kammen DM. 2010. Effects of US maize ethanol on global land use and greenhouse gas emissions: estimating market-mediated responses. *BioScience* 60(3): 223–31

Hertwich E. 2014. Understanding the climate mitigation benefits of product systems: comment on "using attributional life cycle assessment to estimate climate-change mitigation..." *J. Ind. Ecol.* 18:464–65

Høgaas Eide M, Ohlsson T. 1998. A comparison of two different approaches to inventory analysis of dairies. *Int. J. Life Cycle Assess.* 3(4):209–15

Hunt RG, Sellers JD, Franklin WE. 1992. Resource and environmental profile analysis: a life cycle environmental assessment for products and procedures. *Environ. Impact Assess. Rev.* 12(3):245–69

ISO (Int. Organ. Stand.). 2006a. *ISO 14040: Environmental management–life cycle assessment: principles and framework*. Rep., ISO, Geneva

ISO (Int. Organ. Stand.). 2006b. *ISO 14044: Environmental management–life cycle assessment: requirements and guidelines*. Rep., ISO, Geneva

Jaramillo P, Griffin WM, Matthews HS. 2007. Comparative life-cycle air emissions of coal, domestic natural gas, LNG, and SNG for electricity generation. *Environ. Sci. Technol.* 41(17):6290–96

Joseph T, Baah K, Jahanfar A, Dubey B. 2015. A comparative life cycle assessment of conventional hand dryer and roll paper towel as hand drying methods. *Sci. Total Environ.* 515–16:109–17

Karl H, Orwat C. 1999. Economic aspects of environmental labelling. *Int. Yearb. Environ. Resour. Econ.* 2000:107–70

Kendall A, Yuan J. 2013. Comparing life cycle assessments of different biofuel options. *Curr. Opin. Chem. Biol.* 17(3):439–43

Khanna M, Crago CL. 2012. Measuring indirect land use change with biofuels: implications for policy. *Annu. Rev. Resour. Econ.* 4:161–84

Kløverpris J, Wenzel H, Nielsen PH. 2008. Life cycle inventory modelling of land use induced by crop consumption part 1: conceptual analysis and methodological proposal. *Int. J. Life Cycle Assess.* 13:13–21

Knaggård Å. 2014. What do policy-makers do with scientific uncertainty? The incremental character of Swedish climate change policy-making. *Policy Stud*. 35(1):22–39

Krutilla K. 2010. Transaction costs and environmental policy: an assessment framework and literature review. *Int. Rev. Environ. Resour. Econ.* 4:261–354

Laurin L, Amor B, Bachmann TM, Bare J, Koffler C, et al. 2016. Life cycle assessment capacity roadmap (section 1): decision-making support using LCA. *Int. J. Life Cycle Assess.* 21(4):443–47

Lemoine D. 2016. Escape from third-best: rating emissions for intensity standards. *Environ. Resour. Econ*. In press. **https://doi.org/10.1007/s10640–016–0006–6**

Levinson A. 2009. Offshoring pollution: Is the United States increasingly importing polluting goods? *Rev. Environ. Econ. Policy* 4(1):63–83

Lloyd SM, Ries R. 2007. Characterizing, propagating, and analyzing uncertainty in life-cycle assessment: a survey of quantitative approaches. *J. Ind. Ecol.* 11(1):161–79

Luk J, Pourbafrani M, Saville BA, Maclean HL. 2013. Ethanol or bioelectricity? Life cycle assessment of lignocellulosic bioenergy use in light-duty vehicles. *Environ. Sci. Technol.* 47(18):10676–84

Matthews HS, Hendrickson CT, Matthews DH. 2015. *Life Cycle Assessment: Quantitative Approaches for Decisions That Matter*. **http://www.lcatextbook.com**

McAusland C, Najjar N. 2015. Carbon footprint taxes. *Environ. Resour. Econ.* 61(1):37–70

McManus MC, Taylor CM, Mohr A, Whittaker C, Scown CD, et al. 2015. Challenge clusters facing LCA in environmental decision-making-what we can learn from biofuels. *Int. J. Life Cycle Assess.* 20(10):1399–414

Metcalf GE, Weisbach D. 2009. The design of a carbon tax. *Harvard Environ. Law Rev.* 33(2):499–556

Mullins KA, Griffin WM, Matthews HS. 2011. Policy implications of uncertainty in modeled life-cycle greenhouse gas emissions of biofuels. *Environ. Sci. Technol.* 45(1):132–38

Ness B, Urbel-Piirsalu E, Anderberg S, Olsson L. 2007. Categorising tools for sustainability assessment. *Ecol. Econ.* 60:498–508

Ney RA, Schnoor JL. 2002. Incremental life cycle analysis: using uncertainty analysis to frame greenhouse gas balances from bioenergy systems for emission trading. *Biomass Bioenergy* 22(4):257–69

Niederl-Schmidinger A, Narodoslawsky M. 2008. Life cycle assessment as an engineer's tool? *J. Clean. Prod.* 16(2):245–52

Nyquist S. 2003. The legislation of environmental disclosures in three Nordic countries: a comparison. *Bus. Strateg. Environ.* 12(1):12–25

Plevin RJ, Delucchi MA, Creutzig F. 2014a. Response to comments on "using attributional life cycle assessment to estimate climate-change mitigation. . ." *J. Ind. Ecol.* 18(1):468–70

Plevin RJ, Delucchi MA, Creutzig F. 2014b. Using attributional life cycle assessment to estimate climate-change mitigation benefits misleads policy makers. *J. Ind. Ecol.* 18(1):73–83

Portney PR. 1993. The price is right: making use of life cycle analyses. *Issues Sci. Technol.* 10(2):69–75

Portney PR. 2012. *The role of life cycle assessment in environmental policymaking*. Rep., Expert Group Environ. Stud. 2012:4, Gov. Off. Swed., Min. Finance, Stockholm

Rajagopal D. 2013. The fuel market effects of biofuel policies and implications for regulations based on lifecycle emissions. *Environ. Res. Lett.* 8(2):024013

Rajagopal D. 2014. Consequential life cycle assessment of policy vulnerability to price effects. *J. Ind. Ecol.* 18(2):164–75

Rajagopal D. 2016a. A step towards a general framework for consequential life cycle assessment. *J. Ind. Ecol.* 21(2):261–71

Rajagopal D. 2016b. A synthesis of unilateral approaches to mitigating emissions leakage under incomplete policies. *Clim. Policy.* In press. **https://doi.org/10.1080/14693062.2016.1150249**

Rajagopal D, Hochman G, Zilberman D. 2011. Indirect fuel use change and the environmental impact of biofuel policies. *Energy Policy* 39(1):228–33

Reap J, Roman F, Duncan S, Bras B. 2008. A survey of unresolved problems in life cycle assessment. *Int. J. Life Cycle Assess.* 13(5):374–88

Reed DL. 2012. *Life-cycle assessment in government policy in the United States*. PhD Thesis, Univ. Tenn., Knoxville

Searchinger T, Heimlich R, Houghton RA, Dong F, Elobeid A, et al. 2008. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. *Science* 319(5867):1238–40

Sedjo RA, Swallow SK. 2002. Voluntary eco-labeling and the price premium. *Land Econ*. 78(2):272–84

Seidel C. 2016. The application of life cycle assessment to public policy development. *Int. J. Life Cycle Assess.* 21(3):337–48

Spatari S, Bagley DM, MacLean HL. 2010. Life cycle evaluation of emerging lignocellulosic ethanol conversion technologies. *Bioresour. Technol.* 101(2):654–67

Steinmann Z, Hauck M, Karuppiah R, Laurenzi I, Huijbregts MAJ. 2014. A methodology for separating uncertainty and variability in the life cycle greenhouse gas emissions of coal-fueled power generation in the USA. *Int. J. Life Cycle Assess.* 19:1146–55

Suh S, Yang Y. 2014. On the uncanny capabilities of consequential LCA. *Int. J. Life Cycle Assess.* 19(6):1179–84

Tillman AM. 2000. Significance of decision-making for LCA methodology. *Environ. Impact Assess. Rev.* 20(1):113–23

Tinbergen J. 1952. *On the Theory of Economic Policy*. Amsterdam: North Holland

Vandenbergh MP, Cohen MA. 2010. Climate change governance: boundaries and leakage. *NYU Environ. Law J.* 18:221–92

Venkatesh A, Jaramillo P, Griffin WM, Matthews HS. 2011. Uncertainty in life cycle greenhouse gas emissions from United States natural gas end-uses and its effects on policy. *Environ. Sci. Technol.* 45(19):8182–89

Walls M. 2006. *Extended producer responsibility and product design: economic theory and selected case studies*. Disc. Pap. 06-08, Resour. Future, Washington, DC

Walls M, Palmer K. 2001. Upstream pollution, downstream waste disposal, and the design of comprehensive environmental policies. *J. Environ. Econ. Manag.* 41(1):94–108

Wang M, Huo H, Arora S. 2011. Methods of dealing with co-products of biofuels in life-cycle analysis and consequent results within the U.S. context. *Energy Policy* 39:5726–36

Wardenaar T, Van Ruijven T, Beltran AM, Vad K, Guinée J, Heijungs R. 2012. Differences between LCA for analysis and LCA for policy: a case study on the consequences of allocation choices in bio-energy policies. *Int. J. Life Cycle Assess.* 17(8):1059–67

Williams ED, Weber CL, Hawkins TR. 2009. Hybrid framework for managing uncertainty in life cycle inventories. *J. Ind. Ecol.* 13(6):928–44

Zilberman D, Hochman G, Rajagopal D. 2011. On the inclusion of indirect land use in biofuel. *Univ. Ill. Law Rev.* 2:413–34

Zilberman D, Zhao J, Heiman A. 2012. Adoption versus adaptation, with emphasis on climate change. *Annu. Rev. Resour. Econ.* 4:27–53