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Using Life Cycle Assessment to Evaluate the Sustainability of British Columbia's Forest-based Bioeconomy

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British Columbia

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ABSTRACT

There is a growing interest world-wide in an economy more firmly based on bioenergy and bioproducts (i.e., the bioeconomy). Life Cycle Assessment (LCA) is the standard approach for evaluating environmental effects of bioeconomic activities, and therefore is a key component of product certification, market acceptance, and policy development. Given the importance of LCA as an evaluation tool, we provide an overview of LCA principles and methodology with respect to the wood-based bioeconomy in British Columbia, Canada and discuss the evolving efforts within the LCA community to address questions of carbon footprints, land-use change, soil productivity, and biodiversity. Considering the integration among global, national, and local bioeconomies, we conclude that opportunities to benefit from British Columbia's significant biomass resource would be furthered if LCA approaches are developed and incorporated into planning, investment, and decision making. Furthermore, British Columbia's highly regarded forest management regime provides a starting point for evaluating sustainability, but additional information would be needed to carry out the types of assessments that are needed within LCA. A co-ordinated effort by government, academia, and industries who participate in the bioeconomy to explore life cycle thinking would encourage the development of expertise with LCA techniques and improve Life Cycle Inventory databases.

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INTRODUCTION

An expanded bioeconomy based on forest biomass has the potential to provide new economic opportunities while mitigating climate-change impacts through use of this renewable feedstock. Bioenergy is currently the most important bioproduct made from forest biomass. Globally, about 0.7% of the primary energy supply in 2008 came from forest harvesting residues, black liquor from the pulp industry, and residue from other wood industries within industrialized countries (Chum et al. 2011). However, bioenergy production is more prevalent in some countries, and supplies 34% of domestic energy used in Sweden (Hektor et al. 2014). In Canada, about 4% of the total potential energy supply was estimated to come from solid biofuels and charcoal in 2013 (IEA 2014), largely from forest biomass. Much of Canada's current bioenergy production takes place within forest product mills, where more than 95% of wood waste is used for energy production (ICFPA 2015).

Bioenergy produced using biomass from managed forests emits carbon to the atmosphere that is then reabsorbed by the new forests, reducing net carbon fluxes to the atmosphere as compared to fossil fuels. However, the time-scales over which carbon is taken back up by forests (e.g., Ter-Mikaelian et al. 2014), compared with a no-harvesting scenario (Ter-Mikaelian et al. 2015), must be considered. There is disagreement about whether bioenergy is carbon neutral (Berndes et al. 2011) or not (Schulze et al. 2012). There are also concerns about the extent to which changes in ecosystem carbon levels and other aspects of land use affect the benefits of bioenergy (IEA 2013). The benefits of an expanded bioeconomy based on increased biomass removals from forests could be increased if carbon-use efficiency is maximized, and ecosystem damage, loss of biodiversity, and risks to human health (Evans et al. 2010; Petrov 2012) are evaluated and minimized. Governments can play a significant role in the bioeconomy through policies and incentives that encourage sustainable development of this sector (see references in Roach and Berch 2014), but transparent and reliable information is needed so that beneficial activities can be encouraged while investment in activities with unacceptable environmental impacts are avoided.

There is potential for a thriving bioeconomy in British Columbia, Canada, based on available forest biomass. Forests cover 55 million hectares or 58% of the province. Before British Columbia's recent pine beetle infestation, the long-term sustainable harvest level was nearly 70 million cubic metres per year (Province of British Columbia 2010).[1] British Columbia's forests could supply a large amount of forest feedstock for new bioproducts, of which bioenergy is the largest sector within the provincial bioeconomy (Province of British Columbia 2012). Approximately one-third of the fibre (9.4 Mt) in the annual timber harvest is currently used to produce 118 petajoules of thermal and electrical energy for mills and the provincial grid, which is equivalent to 10% of British Columbia's energy demand (Dymond and Kamp 2014). Most of the feedstock is derived from mill residues and only 5% of current production is based on forest harvesting residues (Dymond and Kamp 2014). Bioenergy production could be greatly increased if it was economically feasible and environmentally

1 It is estimated that the timber supply will be reduced by 15–20 million cubic metres between now and 2060, creating stiff competition among existing users let alone new bioeconomy-related ones.

sustainable to recover more harvesting residue (7.8 Mt), trees killed by fire and insects (23.7 Mt), and/or trees that would otherwise be lost through stand-break-up and self-thinning (89.5 Mt) (Dymond et al. 2010).

A number of governance and market-driven systems have evolved to ensure that forests are managed sustainably and that products made from them are produced responsibly. For sustainable forest management (SFM) in Canada, these include government regulations (e.g., Province of British Columbia 2002) and guidelines (e.g., Province of British Columbia 1999), third-party market-driven certification (e.g., Cashore et al. 2004), and standards set by foreign governments for products that they import (e.g., DECC 2014); for production processes, these include emission and environmental regulations (e.g., Province of British Columbia 2003) and market processes (e.g., chlorine-free paper) (CFPA 2015). These sustainability mechanisms and tools typically focus on only one stage of product development or disposal.

In contrast, the "cradle-to-grave" concept originating in evaluations of consumer products in the 1970s (Guinée et al. 2011) gave rise to life cycle assessment (LCA), which is "the compilation and evaluation of the inputs, outputs and potential environmental impacts of a product system throughout its life cycle" (ISO 2006a). Life cycle assessment evolved as a structured and repeatable way of evaluating the environmental impacts of product creation, including bioenergy (Bird et al. 2011), and its strength lies in its analytical consideration of the entire supply chain. All processes are evaluated for their impacts on the environment, and total overall impacts are summed; situations can be identified where decisions simply shift impacts between stages in the production process, or from one environmental problem to another (including disposal). Initially LCAs focussed on greenhouse gas (GHG) emissions, but the environmental considerations have broadened. Life cycle assessment has been widely adopted and currently provides a structured set of core principles for evaluating a range of environmental impacts for many products, services, and industrial activities.

Life cycle assessments improve processes, strengthen market acceptance of products, and can be used to evaluate policy alternatives (Molina-Murillo and Smith 2009). A survey of 190 European companies showed that the most important drivers for conducting LCAs were "product-related environmental problems, cost-saving opportunities, emerging green markets, decision of management, and perceived environmental discussions" (Frankl and Rubik 1999). Within the bioenergy sector, life cycle assessment has played a key role in the development of renewable fuel policies in the United States (Baral 2009).

The structured approach, transparent documentation of process flows, and comprehensive evaluation of environmental impacts in LCA provides an ideal mechanism for informing decision making. Numerous operational guidebooks describe in detail what LCA is and how to apply it (e.g., Guinée et al. 2002; SAIC 2006; Horne et al. 2009; Gaudreault et al. 2015; Schweinle et al. 2015). The core principles and framework of LCA are also described in the International Organization for Standardization (ISO) standard 14040 (ISO 2006a), and requirements and guidelines for LCA are provided in ISO 14044 (ISO 2006b). The results of LCA can be complex to interpret, but there are also now processes for capturing key LCA findings in simplified environmental product declarations (EPDs) (ISO 2006c; Trusty 2012) that are easier for consumers to understand. Examples of EPDs can be found in centralized registries (e.g., EPD Registry

2015, International EPD® System 2015), and these are increasingly being developed for Canadian forest products (e.g., FPInnovations 2013a, 2013b).

The traditional ISO-LCA framework describes an "attributional" approach to LCA, where the goal is to provide a description of environmentally relevant flows of past, present, or future product systems. The "consequential" approach to LCA was developed to consider how, for example, policy decisions affect these environmental flows. A brief review of these two approaches can be found in Gaudreault et al. (2015). New approaches for performing specific analyses for LCA are continually evolving. For example, the Publically Available Specification (PAS) 2050 (BSI Group 2011) sets out specific methods for evaluating certain aspects of carbon accounting related to GHG impacts within attributional LCAs. New methods have also been developed or are being proposed to expand the traditional suite of environmental impacts evaluated in LCA, and the ISO-LCA framework has been instrumental in developing an extensive body of available information to guide new LCA projects.

Life cycle assessment can be used on its own, or can be used to complement and strengthen other sustainability assessment protocols, such as environmental assessments (Finnveden and Moberg 2005). The extra level of quantitative detail that can be added through LCA may improve the accuracy of impact assessment for a particular environmental attribute or value, and can also be used to inform strategic environmental assessments or other aspects of policy (Manuilova et al. 2009). In British Columbia, environmental assessment is a proponent-driven, project-specific process through which proposed projects require the submission and acceptance of information on their environmental and other impacts (Province of British Columbia 2011). Despite the potential application of LCA as a complementary analysis to environmental assessment, it may not be suitable for evaluating the wide range of local issues at the detail required for specific projects, and so LCA cannot generally be considered as a replacement for environmental assessments.

There has been a movement in recent years towards incorporating the environmental impact results from LCA, economic factors from life cycle costing models, and social impacts derived from a social LCA into an all-encompassing life cycle sustainability analysis (Guinée et al. 2011). The need for integration arises because economic effects and behaviours outside the scope of LCA can directly or indirectly affect interpretations in decision-making processes (Elghali 2002). Life cycle sustainability analysis is envisioned as a more comprehensive framework for integrating results from an array of models so that all three pillars of sustainability (environmental, economic, and social) can be evaluated.

Given the emerging importance of LCA as an evaluation tool, and anticipating its possible application by governments in policy making and by markets through EPDs relevant to the bioeconomy in general, and biofuels and bioenergy production in particular, the purposes of this report are to (1) provide land managers with a brief introduction to LCA and its potential use for evaluating the environmental impacts of wood-based bioenergy in British Columbia; (2) describe specific aspects of forest land management in British Columbia as they relate to the use of LCA for evaluating environmental impacts within the bioeconomy; and (3) outline information gaps and research needs that could be addressed to support the use of LCA in the development of British Columbia's bioeconomy.

The four key components of an environmental LCA based on ISO 14040 (ISO 2006a, 2006b) are (1) *goal and scope definition,* (2) *inventory analysis,* (3) *impact assessment,* and (4) *interpretation* (Figure 1). The goal and scope definition stage is where initial choices are made regarding the specific question(s) to be addressed; the target audience; the technological, geographical, and temporal scope (i.e., system boundary); the impact categories

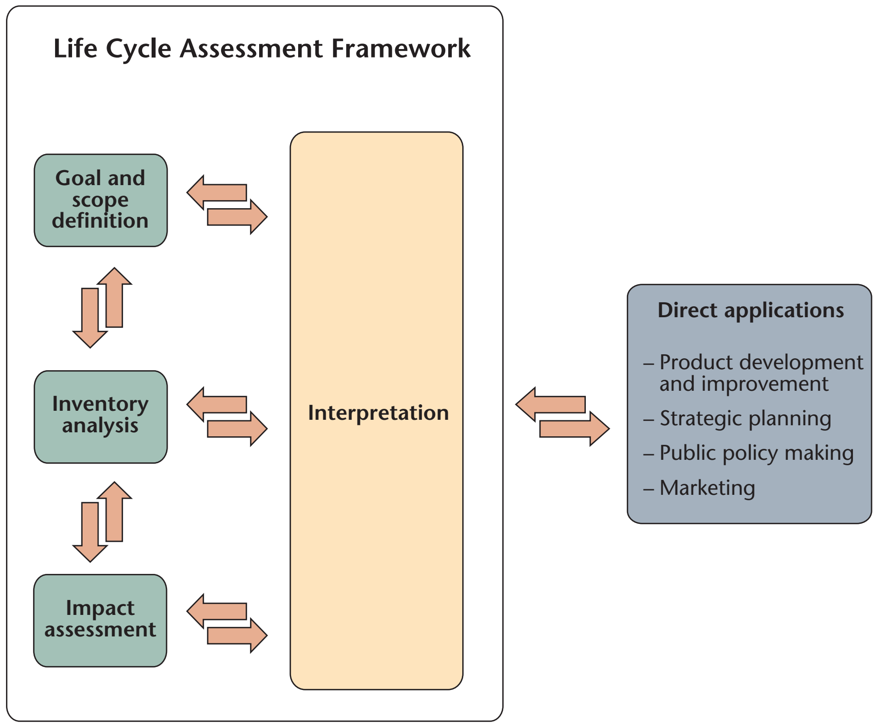

FIGURE 1 *Stages of an LCA (ISO 2006a).*

to be assessed; and the products of the LCA (Guinée et al. 2002). For example, an assessment of forest-based bioenergy in British Columbia could focus on three wood-based biofuel systems (hog fuel, pellets, and cellulosic ethanol) based on woody biomass from forest operations, logging and sawmill residues, mountain pine beetle rehabilitation, and short-rotation intensive culture (SRIC). One of the stated principles of ISO 14040 is an "environmental" focus. LCAs that adhere to this standard (i.e., ISO-LCAs) have typically addressed selected environmental impacts of a product system but economic and social aspects have been considered beyond their scope (Finkbeiner et al. 2006). The need for integration of economic and social factors is driving new approaches that are more inclusive. Decisions made at the *goal and scope definition* stage determine the structure of the LCA study, the functional unit,[2] and the methods that will be employed to achieve the stated goal(s). In LCA, all environmental inputs and outputs (i.e., flows) are normalized to a

2 A functional unit is the quantified performance of a product system for use as a reference unit for the analysis and for making comparisons against alternative products that perform the same function.

functional unit, usually defined as one megajoule (MJ) or gigajoule (GJ) of the relevant biofuel energy content, to facilitate comparisons across and between process domains such as production and the environment.

The energy and raw material requirements, emissions, and other environmental releases over the product life cycle are then quantified in the *inventory analysis or life cycle inventory* (LCI) stage (SAIC 2006), which results in a quantitative depiction of flows of material and energy to and from the environment that result from the production processes. A key part of the LCI is production of a flow diagram, which identifies all of the processes that cause significant flows to the environment, and defines the system boundary (Figure 2). Data for the LCI are available through various public (e.g., U.S. LCI Database 2015) and commercial (e.g., ecoinvent [Weidema et al. 2013]) online databases if local data are not available. Life cycle databases are also being compiled that are specific to Canada (CIRAIG 2015) and include data relevant to the forest bioeconomy.

The LCI stage also specifies the methods used to allocate flows through which the products from one process then become part of another process (e.g., when recycling is part of a process). The choice of allocation method is one of many decisions that need to be taken at this stage. Using different allocation approaches within a bioelectricity supply chain, for example, can lead to a 16–66% variation in GHG impacts compared to a reference system. Although there may not be an objectively "correct" approach, LCAs informing bioenergy policy would likely be improved by using physical partitioning based on energy content (Wardenaar et al. 2012). The exclusion of insignificant flows (cutoffs, thresholds), the level of detail required (specificity), the treatment of missing data, and other assumptions made in this stage also significantly affect LCA outcomes. The LCI might also reveal data limitations or other issues that force a review of study goals and scope. The cause of different outcomes from LCAs that appear to address the same question can often be traced back to methods employed and assumptions made at the LCI stage. The requirement to document decisions made at all stages of an LCA is therefore a major strength of the ISO-LCA approach. This transparency facilitates interpretation in the context of the methodologies and assumptions made, and helps the wider LCA community to better understand the LCA outcomes.

The environmental flows determined in the LCI stage are assigned to impact categories in the *life cycle impact assessment* (LCIA) stage; the level of environmental impacts are then determined by applying *characterization factors* to specify the relationships between environmental flows such as GHG emissions and resultant impacts. A variety of approaches and tools exist for determining the environmental impacts of environmental flows and, as with LCI, the decisions and assumptions incorporated into the LCIA affect overall outcomes. TRACI 2.1 is an impact assessment methodology developed by the U.S. Environmental Protection Agency (Bare 2012) for evaluating impact categories, and can be applied to bioenergy production in British Columbia (Table 1; after Mahalle et al. 2013). TRACI 2.1 provides characterization factors to estimate potential impacts of GHGs, air and water pollution, human health, and resource depletion of fossil fuels. Because an LCA study is intended for use in public policy formulation, the impacts should be selected based on the common concerns regarding the use of wood-based biofuel and the breadth of life cycle resource and materials inputs and releases to air, water, and land. Though they

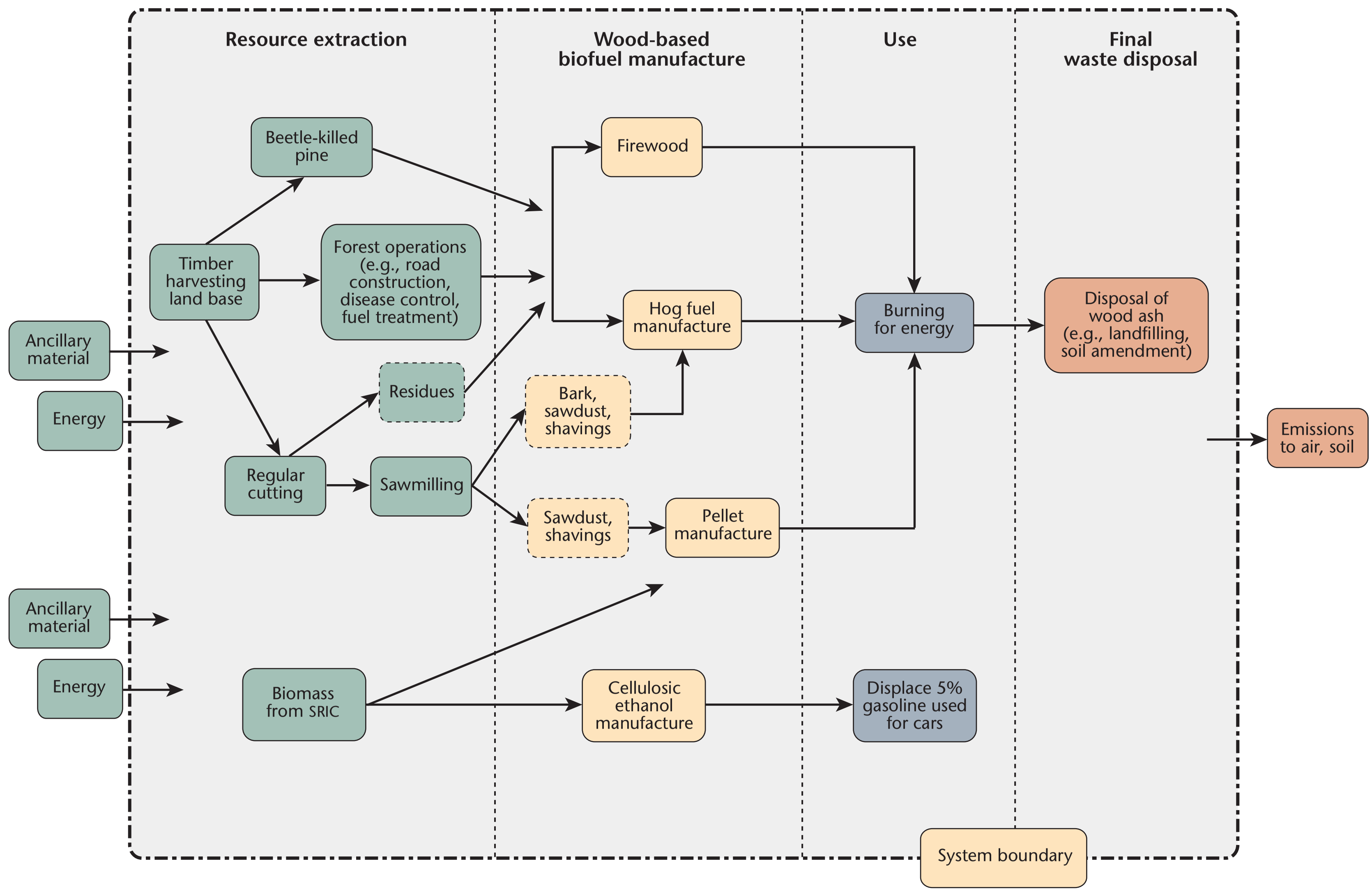

FIGURE 2 *Theoretical system boundary and process flows for a hog fuel production system in British Columbia (reproduced from Mahalle et al. 2013).*

TABLE 1 *Impact categories, indicators, and inventory requirements applicable to bioenergy in British Columbia (after TRACI 2.1)*

Impact category	Category description	Required inventory data	Impact category indicators	Applicability to bioenergy in British Columbia
Acidification	Acidification is the increasing concentration of hydrogen ion (H^+) concentration in the local environment.	Mass of air emissions such as sulphur oxides (SO_x), nitrogen oxides (NO_x), ammonia (NH_3), hydrogen chloride (HCl).	**Acidification Potential** is calculated on the basis of its moles H^+ ion equivalence produced per functional unit.	Compared to coal or many other fuels, wood biomass combustion typically produces low amounts of SO_x. There may be an issue with kraft mill sludge burning (Washington State Department of Ecology 2003; Beauchemin and Tampier 2008).
Global Climate Change	Global warming is an average increase in the temperature of the atmosphere near the Earth's surface and in the troposphere, which can contribute to changes in global climate patterns.	Mass of GHG emissions carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4).	**Global Warming Potential** (GWP) kg or tonnes CO_2 equivalent produced per functional unit.	CO_2 is released from combustion of fuels and from land use, at least in the short term (Searchinger et al. 2009).
Human Health: air pollutants	Particulate matter of various sizes (PM_{10} and $PM_{2.5}$) has a considerable impact on human health.	Mass (kg) released of: particulate matter (PM_{10}, $PM_{2.5}$), nitrogen oxides (NO_x), sulphur dioxide (SO_2), total suspended particulates (TSP).	**Human Health Potential** (Respiratory Effects) is calculated on a kg $PM_{2.5}$ equivalent produced per functional unit.	Particulates are an important environmental output of biomass combustion and need to be traced and addressed (Environment and Human Health Inc. 2010).
Human Health: cancer, non-cancer, and ecotoxicity	Chemicals released into the environment can cause detrimental effects on human health and ecosystems.	Mass of chemicals and metals released to air, water, and soil along the product chain.	**Carcinogenicity Potential** – an equivalent mass of benzene (CTUh).[a] **Human Health Potential** (Non-cancer) – an equivalent mass of toluene basis, in comparative toxic units (CTUh). **Ecotoxicity Potential** – an equivalent quantity of 2,4-D in comparative toxic units (CTUe).[b]	Trace metals and heavy metals that cause human health impacts are a concern with wood smoke and ash (Patterson 2001; Environment and Human Health Inc. 2010).
Smog Formation	Air emissions from industry become trapped at ground level, and produce photochemical smog in the presence of sunlight.	Mass of volatile organic compounds (VOCs) and acidification emissions produced along the product chain.	**Photochemical Ozone Formation Potential** (Smog) – expressed on a mass of equivalent NO_x basis produced per functional unit.	VOCs, NO_x are emitted from biomass burning and are a concern in some municipalities (e.g., Metro Vancouver 2011).
Land Use[c]	Land use reflects impacts to soils, biodiversity, and other environmental categories not included elsewhere.	Potential indicators for: soil: SOM, disturbance area biodiversity: species richness, occupied area.	TRACI 2.1 does not provide characterization factors to calculate indicator results.	Potential for soil degradation, ecosystem damage, or endangered species to be negatively affected.

a Comparative Toxic Units, human.

b Comparative Toxic Units, environment. 2,4-D = 2,4-Dichlorophenoxyacetic acid.

c Land-use impacts are not specifically addressed within TRACI, but are included here to illustrate their effect on bioenergy assessments in British Columbia.

are a primary management and policy concern in British Columbia, land-use impacts to values such as biodiversity and soil quality are not currently included in TRACI 2.1. However, it is anticipated that future versions of TRACI will be able to address potential impacts of land and water use (Bare 2012).

Results are analyzed, conclusions are drawn, and limitations of the study are documented in the *interpretation* stage of LCA. This stage includes sensitivity analysis, which can identify the data needed to make findings more robust. As in previous stages, transparency is essential at the interpretation stage so that an independent reviewer can assess results in the context of the study approach used.

EVALUATING WOOD-BASED BIOENERGY IN BRITISH COLUMBIA USING LCA

A number of traditional LCAs have been done on bioenergy products in British Columbia. As is typical of traditional LCAs, only fossil-fuel emissions were considered in these studies and forest carbon dynamics, soil productivity, and biodiversity were ignored. These LCAs include the examination of wood pellets exported to Europe; pellets versus coal for electricity; and pellets compared to firewood heat, natural gas heat, sewer heat, and ground heat. The production of wood pellets is a growing part of the forest-based bioeconomy in British Columbia. Life cycle assessment has been used to analyze the supply chain of pellet production, transportation, and use by applying a streamlined LCA to evaluate the carbon footprint of British Columbia wood pellets exported to Europe along a six-step supply chain (Magelli et al. 2008; Pa et al. 2012). The LCA results in these two studies suggest that enhanced GHG benefits could be derived by (1) reducing emissions caused by marine transportation by increasing use of wood pellets for domestic heating in British Columbia, and (2) improving energy efficiency in harvesting operations and pellet production.

A more recent study took into account the loss and regrowth of forests (forest carbon dynamics) through harvesting in assessing pellet LCA where the pellets were used in the Netherlands to replace coal for electricity generation. In this example, the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3 [Kurz et al. 2009]) was used to evaluate stand- and landscape-level forest carbon for (1) no harvest, (2) pellets from sawdust, (3) pellets from slash, and (4) pellets from dead trees in pine- and spruce-dominated forests in British Columbia's interior. This approach identified scenarios with short break-even times (0–20 years when pellets from pine slash replace coal), and longer times (20–39 years when slash is burned and only sawdust is used for pellets). Most importantly, there was no benefit indicated of replacing coal with pellets when harvesting stands exclusively for bioenergy, a conclusion that would not have been reached without considering the forest carbon. The authors also carried out sensitivity analysis to identify information needs (e.g., rates of wood decomposition in landfills) that could support more robust conclusions (Lamers et al. 2013).

Life cycle assessment has also been used in British Columbia to evaluate the potential benefits of using wood pellets for residential and district heating systems (Ghafghazi et al. 2011; Pa et al. 2013). Compared to firewood, pellets had higher "upstream"[3] emissions primarily because of the electricity used in sawmills and pellet mills; but "downstream" emissions were much lower because pellet stoves are more efficient than fireplaces, and pellets have lower water content than typical firewood. The same study illustrated the economic savings and increased benefits to human health because of reduced emissions if low-efficiency fireplaces were replaced with high-efficiency pellet stoves (Pa et al. 2013). The environmental burden of using wood pellets, natural gas heat, sewer heat, or ground heat for a district heating system base-loads were evaluated by Ghafghazi et al. (2011). This study showed that GHG impacts were less for ground-source heat pumps than wood pellet burners in Vancouver, British Columbia, primarily because of British Columbia's low-carbon electricity generating profile (i.e., high proportion of hydropower).[4] Wood pellets had lower GHG, aquatic ecotoxicity, and non-renewable energy impacts than natural gas, but higher impacts for respiratory inorganics and terrestrial ecotoxicity. Both of these British Columbia studies illustrate the value of LCA for comparing alternative energy sources.

There are not yet LCAs that quantify the potential environmental benefits and impacts associated with using forest residues to generate electricity in British Columbia, but studies conducted elsewhere may have some application. For example, a wide-ranging review of 94 bioenergy LCAs by Cherubini and Strømann (2011) found that bioenergy systems generally have lower efficiencies than conventional/fossil energy systems. This may be particularly relevant to British Columbia where most electricity is generated by hydropower. Most studies concluded that net GHG emissions were significantly lower for electricity generated from biomass when the biomass was derived from low-input sources and waste streams, although there was substantial variation in results partly because of differences in methodological approaches used. Finally, in bioenergy studies that examined life cycle consequences on human and ecosystem toxicity as well as on other impact categories, bioenergy systems focussed on intensive agricultural practices coupled with use of fertilizers often had greater environmental impacts than fossil reference systems.

Results from studies conducted elsewhere can provide general information relevant to British Columbia but do not address the characteristics of British Columbia's woody biomass resource; energy infrastructure; or environmental, economic, and social considerations that are unique to British Columbia and which affect the environmental benefits and burdens associated with British Columbia's bioenergy sector. The data from elsewhere are not representative of the conditions in British Columbia as well due to differences in technology, transportation distances and modes, and electricity grid make up of energy sources. Electricity in British Columbia is generated mostly using hydropower compared to the electricity produced in the United States, which is mainly fossil fuel–based (coal). Therefore, bioenergy produced using British Columbia electricity has a better environmental profile compared to the same biofuel manufactured using U.S. electricity. Different combustion technologies can result in different emission amounts. Gasification is more efficient and produces fewer emissions compared to conventional wood boilers (Beauchemin and Tampier 2008), for instance. As a result, British Columbia would need to develop its own LCI databases to inform policy decisions with regard to its

3 Upstream refers to production and transmission phases. Downstream refers to the utilization phase.

4 Had the study taken into account the impacts of dam and power system construction on biodiversity and soil productivity, perhaps these results would have differed.

bioenergy sector. This need was addressed in a review of currently available options for a broad-based evaluation of bioenergy in British Columbia that guided the development of an operationally feasible approach for applying LCA to bioenergy development in British Columbia (Mahalle et al. 2013). Tools and data available for evaluating impacts on energy, GHGs, land use, and ecosystem quality (including soil and biodiversity) were considered, and it was concluded that an extended life cycle sustainability assessment for bioenergy in British Columbia could be developed with ISO-LCA to address the environmental component, but land-use, economic, and social impacts would need to be evaluated as separate elements.

LCA AS AN EVOLVING APPROACH FOR EVALUATING ENVIRONMENTAL IMPACTS OF LAND USE AND LAND-USE CHANGE

Life cycle assessment is an evolving concept and approach that incorporates a wide variety of methodologies. Three different periods of development of LCA have been described (Guinée et al. 2011): (1) the conception period (1970–1990), when there was no agreed-upon theoretical framework and hence a wide variety of approaches were used; (2) the standardization period (1990–2000), when co-ordinated scientific activity led to development of a consistent LCA framework, including specific terminology, methods, and standards; and (3) the elaboration period (2000–2010), when implementation of policies supported by LCA increased. Looking to the future, new and robust methods are needed to inform increasingly stringent and specific policy questions, and to address theoretical limitations in the LCA approach itself.

There are at least two key challenges: (1) how to incorporate land-use impacts to ecosystems services from feedstock management practices (e.g., soils, biodiversity, and water) (see reviews by Gaudreault et al. 2015; Schweinle et al. 2015) and, if it takes place, land-use change (e.g., Sánchez et al. 2012); and (2) how to address assumptions about how a particular system can affect and be affected by the wider economic and social context (e.g., Rajagapol et al. 2011). These and other issues continue to challenge LCA practitioners and provide an impetus for further study and methodology development. Greenhouse gas accounting is a core of traditional LCA, but applications to woody biomass–based bioenergy are complex because of the land management aspects of feedstock production that need to be considered, including forest carbon cycling and land-use change. Methods for incorporating forest carbon cycling into LCA have progressed in recent years, but work on land-use impact categories in general is arguably at the conception stage of development (Guinée et al. 2011).

In LCA, the physical impacts of biomass removals through forestry operations and forest management can be considered to result from land occupation. Land occupation is the postponement of land recovery to its "natural vegetation state" (Schmidt 2008). Sites are disturbed and re-disturbed through harvesting, allowing for recovery through stand establishment and growth between disturbance events (Weidema and Lindeijer 2001; Doka et al. 2006; Nolan et al. 2009). The impacts of land occupation to values such as biodiversity and soil productivity can be considered a function of the area of land affected, the duration of the occupation, and changes in the quality of the land during occupation, expressed relative to the functional unit of production (Milà i Canals et al. 2007b; Nolan et al. 2009). In reality, forest management is a combination of occupational and transformational impacts (Willskytt 2015), with harvesting initially transforming the land, after which the site regrows over the next rotation (relaxation time) back to a mature forest (natural vegetation state).

Forest Carbon, Land-use Change, and GHGS

Many LCAs consider that emissions from woody biomass are inherently carbon neutral because the carbon emitted to the atmosphere at energy conversion is taken back up through the growth of the next rotation. This may, however, lead to flawed conclusions in LCA because the potential for bioenergy to reduce GHGs depends inherently on (1) the specific source of the biomass, (2) whether the growth and harvesting of the biomass for energy captures carbon above and beyond what would have been sequestered without the bioenergy intervention, and (3) whether carbon stock changes on land are considered (Searchinger et al. 2009). Assumptions of carbon neutrality can also arise through misunderstanding of the IPCC accounting rules because emissions at the point of energy conversion are not accounted for within the "Energy Sector"; however, the IPCC recognizes that additional GHGs can be emitted through harvesting and regrowth, land-use changes, processing and transportation of biomass, methane and N_2O emissions from combustion, and use of fertilizers and liming (IPCC 2015).[5]

The issue of carbon stocks on land is partly addressed in PAS 2050 by considering land-use change as a GHG impact (BSI Group 2011). If land is removed from one category (e.g., forest) and placed into another (e.g., cropland) as a result of the bioenergy activity, then a land-use factor is applied to reflect the difference in carbon stocks between the two land-use types, and therefore the GHG impact of land-use change (BSI Group 2011). Carbon stock changes are not included in LCA if the land use remains the same, however, which would typically be the case for bioenergy operations on forest lands in British Columbia.

Similarly, a zero-GHG burden for biomass is proposed when it is residual biomass not traditionally used for energy production (Turconi et al. 2013), such as material that is typically burned in incinerators (e.g., industrial waste) or as part of forest management (e.g., forest harvesting residue in some forest ecosystems). On the other hand, making such assumptions when the fate of woody debris in the absence of bioenergy use is not known could compromise the credibility of conclusions in an LCA.

More detailed evaluation of the effects of bioenergy development on carbon stocks of managed forest lands within LCA would provide important information to guide bioenergy policy and could be accomplished in a number of ways. Helin et al. (2012) reviewed potential approaches for including forest carbon cycling in LCA from four perspectives: (1) treatment of reference land use, (2) timeframe and timing of carbon emissions and sequestration, (3) substitution effects, and (4) indicators. They propose that natural relaxation (regrowth) is an appropriate reference system for attributional LCA, while an alternative land use could be applied in consequential LCA.[6] Dynamic forest modelling can also be used to address the timing of emissions and sequestration, and the climatic indicator should take cumulative radiative forcing into

5 For further discussion of forest carbon accounting, see Bernier et al. (2012), Agostini et al. (2013), Miner and Gaudreault (2013), EPA (2014), Miner et al. (2014), and Ter-Mikaelian et al. (2014).

6 For further discussions of attributional and consequential LCA, see Brander et al. (2009) and Gaudreault et al. (2015).

account. Helin et al. (2012) argue that such evaluations should not include any assumptions about product substitution impacts (i.e., biomass carbon stored in a product) because these are subject to considerable uncertainty.

Tittman and Yeh (2012) also argued that LCAs need to employ a detailed understanding of forest ecosystems as well as industrial ecology, and proposed a comprehensive framework for evaluating GHG impacts that incorporates stand-level dynamics and industrial processes. They suggested that risk management approaches should be considered for GHG impact evaluation because although forests both incrementally sequester and emit GHGs, they also have the potential to release very large pulses of GHGs (e.g., through wildfire). The proposed framework evaluates GHGs as a result of dynamic linkages between (1) wildfire emissions, (2) decomposition, (3) forest products, (4) bioenergy, and (5) displacement of fossil fuels (Tittman and Yeh 2012). Similar considerations were discussed by Lamers et al. (2013), who demonstrated how forest carbon dynamics and results from LCA can be incorporated into evaluations of carbon "break-even" times, such as that for wood pellets produced from forests affected by pine beetles in British Columbia.

Soil Productivity

Soil and related properties help determine the quality of forest ecosystem services including: erosion regulation, water regulation, water purification, biotic (primary) production, and carbon sequestration (Saad et al. 2011). Erosion and water quality in forested areas in British Columbia are protected through regulations and guidelines. The greatest soil-related environmental concern of forest-derived bioenergy in British Columbia is the loss of soil productivity through intensive harvesting and removal of biomass because it contains both nutrients and organic matter.

Soil productivity has long been recognized as important for LCIA, but no approach has yet emerged that can be consistently applied and is recognized as a standard analytical method (Gaudreault et al. 2015; Schweinle et al. 2015). Within forestry, the impacts of land-use change (i.e., land transformation or conversion) and land use (i.e., land occupation) on soil productivity can be considered as functions of (1) area affected, (2) changes in soil quality (based on maps of predicted properties), and (3) the time needed after treatment (i.e., relaxation time) to reach the potential natural vegetation state (Saad et al. 2011, 2013). However, spatial variability in soil properties and productivity (i.e., biotic production potential) plus temporal variability (i.e., time from harvesting) will complicate the development of characterization factors to describe associated environmental impacts (Schweinle et al. 2015). Specific factors may be needed for different soil types, unless generic soil quality indicators can be developed, validated, and applied.

Various indicators have been suggested for measuring impacts on soils in LCIA and how these affect a number of ecosystems services; some of these indicators have been recommended as part of multi-criteria indicators (e.g., soil quality). Individually, these indicators include soil physical properties such as macropore volume, aggregate stability, and rooting depth (Oberholzer et al. 2012); texture and gravel content (Saad et al. 2011, 2013); soil erosion (Cowell and Clift 2000; Saad et al. 2011, 2013; Núñez et al. 2012); soil compaction (Cowell and Clift 2000; Saad et al. 2011, 2013); nutrient capital (Cowell and Clift 2000), cation exchange capacity (Saad et al. 2011, 2013); salts (Cowell and Clift 2000); soil pH (Cowell and Clift 2000; Saad 2011, 2013); pollutants such as heavy metals and organic pollutants (Oberholzer et al. 2012); soil biodiversity such as earthworm biomass, microbial biomass, and microbial activity (Oberholzer et al. 2012); soil carbon (Oberholzer et al. 2012); and soil organic matter (SOM) (Cowell and Clift 2000; Saad et al. 2011, 2013).

Soil quality has been considered in LCIA based on specific land-use impacts on soil erosion, soil fertility, and soil structure (Garrigues et al. 2012), and ecosystem thermodynamics (Wagendorp et al. 2006). Multi-criteria indicators have also been proposed to better capture the complexity of soil quality. Oberholzer et al. (2012) incorporated nine soil property indicators (earthworm biomass, microbial biomass, microbial activity, organic carbon, heavy metals, organic pollutants, macropore volume, aggregate stability, and rooting depth) to assess the impact of agricultural soil use on soil quality in central Europe. In contrast, Milà i Canals et al. (2007a) used only SOM to indicate soil quality, acknowledging, however, that SOM may not directly address impacts related to erosion, compaction, acidification, salinization, and accumulation of toxic substances. Despite such limitations, and recognizing the temporal and inherent spatial variation in soil systems, they considered SOM to be the most practical and relevant indicator for life support functions in most agro-forestry systems studied using LCA, and a better option than no assessment of soil quality.

However, establishing empirical relationships and thresholds for SOM content specifically related to evaluating forest biomass removals for bioenergy in British Columbia will require a nuanced approach as the impacts of harvesting intensity on SOM (and soil carbon) may not be generalizable. For instance, one meta-analysis showed an increase in carbon in the forest floor with stem-only harvesting (especially for conifers but not with hardwoods) and a slight decrease with whole-tree logging, but there were no effects in the mineral soil (Johnson and Curtis 2001). A second meta-analysis showed significant effects of forest harvesting on soil carbon despite the high levels of spatial and temporal variability in forest soil carbon measurements (Nave et al. 2010). A third review found that no general conclusions could be drawn, especially regarding the longer-term impacts of harvesting intensity on soil carbon (Clarke et al. 2015).

Linking soil carbon to forest site productivity would be an important step in validating the importance of SOM as an indicator in LCIA, but such links may be extremely complex or even impossible to quantify. Temporal as well as spatial variability is thought to be problematic (Clarke et al. 2015), although time since harvesting has not yet been shown to have a consistent effect on soil C recovery in meta-analyses (Nave et al. 2010) and reviews (Thiffault et al. 2011; Clarke et al. 2015).

In some situations, seedling establishment and at least the early stages of forest regrowth were shown to be relatively insensitive to organic matter removal (Ponder et al. 2012), perhaps because factors other than those related to SOM limited growth at this stage of stand development. In contrast, an Ontario study found a very strong relationship between total soil carbon content (to 20 cm in the mineral soil) and 5-year tree height growth increment (10–15 years) for jack pine, but not for black spruce (Hazlett et al. 2014). In a third study, a significant relationship was found between above-ground growth (mass) and a risk rating based on soil carbon concentration in the top 30 cm of the mineral soil for three sites in eastern Canada (Thiffault et al. 2014) but an accompanying scattergram suggests that the relationship is not strong, especially across sites. These findings emphasize the need for a better understanding of species- and site-specific relationships over relevant time periods before applying SOM indicators in bioenergy LCA.

Despite considerable investment in forest soils research over more than three decades, there are currently no universal forest soil indicators for predicting forest management impacts. Existing and new research efforts will likely generate new field data and spatial predictions for soil attributes such as SOM, texture, and pH for large parts of Canada and British Columbia within the next decade (e.g., Mansuy et al. 2014; Thiffault et al. 2014), but the conceptual framework and understanding of how to incorporate soil properties into reliable measures of, for example, productivity will likely remain challenging. A disciplined, strategic, and long-term research commitment is required to incorporate soil impacts of forest management for bioenergy feedstock production into LCA in Canada (e.g., Saad et al. 2011).

Biodiversity

Biodiversity underpins the functions and processes of forest ecosystems and the provision of ecosystem goods and services. Resource use and emissions from forest-derived bioenergy can have an impact on biodiversity through land use (forest management), water use, emission of pollutants, and climate change (Schweinle et al. 2015). Habitat change and habitat loss are key drivers for forest management impacts to biodiversity. The reduction of dead wood is considered to be one of the most severe biodiversity problems in Swedish landscapes managed for high biomass volume removal, along with large-scale transformation of old forest and increased landscape fragmentation (Arn 2013).

As with soil productivity, the biodiversity impacts of initial land transformation and land occupation due to forest management can be considered a function of the area of land, the duration of the occupation, and changes in the biodiversity quality of the land during occupation, expressed relative to the functional unit of production (Milà i Canals et al. 2007b; Nolan et al. 2009). However, forest management regimes producing woody biomass for bioenergy may vary, resulting in variable impacts to biodiversity. Short-rotation plantations result in intensive transformation and occupation impacts, which might require restoration at the end of occupation to facilitate recovery. In contrast, forests managed sustainably according to a natural disturbance paradigm could be expected to have roughly constant ecological qualities, with measures of biodiversity oscillating around some average value throughout multiple forest rotations (Doka et al. 2006). Forest management occupation could, however, result in incremental and accumulated reductions in components of biodiversity, such as dead woody material or species abundance (Tinker and Knight 2001; Muys and García Quijano 2002; Gerzon et al. 2011). The accumulated effects of long-term forestry occupation over multiple harvest rotations may lead to a reduction in the ecological quality and resiliency of a site beyond some threshold, making recovery to the original state impossible; transformation to a qualitatively different biodiversity state would occur (Köellner and Scholz 2007; Campbell et al. 2009; Souza et al. 2015). Furthermore, characterization of forestry land occupation impacts to biodiversity does not consider the cumulative effects of other anthropogenic activities (i.e., land development and resource extraction) and environmental conditions (i.e., climate change) (Souza et al. 2015).

The effective quantification and interpretation of land-use impacts on biodiversity in LCIA are difficult because of methodological and conceptual constraints (Jonsell 2007; de Baan et al. 2013). Indicator selection and interpretation for biodiversity impacts presents the "messy problem" of integrating analytical and interpretive simplicity with the complexity of the system under study. Direct (e.g., species richness, species abundance) and indirect (e.g., fragmentation, habitat supply) indicators of biodiversity may not match the relevant temporal and spatial scales of life cycle inventories, can pose difficulties in the detection and interpretation of change, and are frequently data-poor and reliant on expert opinion (Milà i Canals et al. 2007b; Michelsen 2008; Oyewole 2010). In particular, the development of biodiversity characterization and weighting factors for attributing environmental flows to a functional unit is challenging and controversial (e.g., Köellner 2000), and characterization factors are location- or region-specific and scale-dependent, confounding efforts to develop standardized or global impact characterizations for biodiversity (Oyewole 2010; Souza et al. 2015; Willskytt 2015).

At least 17 biodiversity-related indicators have been suggested for LCIA (Schweinle et al. 2015), including species richness, species-area relationships (including rarefaction curves), and surrogate species distribution models (Souza et al. 2015). The most widespread approach is to include land-use impacts to biodiversity in LCA through characterization factors based on species richness (Willskytt 2015), where impacts are modelled as a loss of species richness due to land-use or conversion relative to a reference state (e.g., natural vegetation state) (Schmidt 2008). Calculation of these characterization factors can be complicated and limited by the availability of usable data (e.g., Michelsen 2008). The choice of reference baseline may vary and may often be derived through value choices (Souza et al. 2015), such as those set during strategic land-use planning initiatives in British Columbia. This emphasis on the compositional aspects of biodiversity (i.e., species richness, assemblages, or accumulation) ignores the functional and population effects of land use on biodiversity (Curran et al. 2011; Souza et al. 2015); appears less sensitive to land use than other indicators (de Baan 2013; Turner et al. 2014); and is not recommended for use in some types of LCA at this point (e.g., garment industry) (Egorova et al. 2014). Species-area relationship models may improve with improvements in data availability and quality, but they will still be limited by high uncertainty arising from the complexity of biodiversity, where aspects such as spatial distribution and genetic, functional, and structural diversity are not considered (Egorova et al. 2014). Correlations in the response of species richness between different taxonomic groups are unclear and inconsistent (Koellner 2000; Michelsen 2008; Schmidt 2008), and specific species and species functions have been found to be more important in global biodiversity hotspot identification than the total number of species (Stuart-Smith et al. 2013 in Willskytt 2015). The list of current shortcomings in biodiversity impact modelling in LCIA is lengthy (e.g., invasive species are not considered) (Curran et al. 2011).

Given the complexity of biodiversity assessment in general, one indicator will not suffice for the assessment of land-use impacts to biodiversity, and several aspects of biodiversity should be considered at both midpoint and endpoint levels in an LCIA (Souza et al. 2015). Research is required to identify the most robust (i.e., sensitive and informative) combination of biodiversity indicators and taxonomic groups, and evaluate how to interpret individual and interactive indicator responses in an LCIA framework (Souza et al. 2015; Willskytt 2015). Given the difficulties of normalization to a functional unit and indicator characterization, it might be easier to evaluate biodiversity separately from LCA (Willskytt 2015), pairing LCA with other land management governance mechanisms (e.g., environmental impact assessments) (Gaudreault et al. 2015; Schweinle et al. 2015). Gaudreault et al. (2015) suggest it is unlikely that LCA will ever be able to adequately quantify site-specific impacts because of the "inherent global and comprehensive nature of LCA."

However, the challenges to identifying a suite of meaningful and representative indicators of forest management impacts to biodiversity are not unique to LCIA. In Sweden, most forest management proponents rely on indirect measures as indicators of biodiversity, including dead wood, old trees, and other structural attributes that speak to the prerequisites for biodiversity rather than biodiversity itself (Arn 2013). Similarly, British Columbia sets minimum coarse-filter biodiversity objectives and targets for structural (e.g., wildlife trees, coarse woody debris) and compositional (e.g., ecological representation) attributes in forest management legislation and regulation (Forest Planning and Practices Regulation, Province of British Columbia 2014; *Forest and Range Practices Act,* Province of British Columbia 2015) to manage for stand- and landscape-level biodiversity. Strategic land-use planning initiatives in British Columbia may also include additional coarse-filter surrogate or focal species (umbrella, keystone, or indicator) in biodiversity conservation (e.g., Holt 2005; Horn et al. 2009). Biodiversity impact assessment in a changing climate is positively evolving and developing in British Columbia, but forest biodiversity evaluations focus on structural attributes, rather than biological and ecological outcomes (Forest and Range Evaluation Program, Province of British Columbia 2009), and a cumulative effects framework is still actively being developed and implemented (Auditor General of British Columbia 2015). Reliance on complementary land management mechanisms in British Columbia may remove the problem from LCA, but the same level of indicator research and development is required to address the challenges facing effective biodiversity impact assessment in any British Columbia assessment framework.

CONCLUSION

British Columbia's natural resource science program and comprehensive protocols for compliance, monitoring, and evaluation provide information on a wide range of environmental values used to guide policy development and forest management operations. Existing soil and biodiversity monitoring programs in British Columbia could inform the process for including these impact categories in LCA (Berch et al. 2012); assessment methodologies for other potentially relevant impact categories (including water, wildlife, cultural heritage, and visual quality) have also been developed in British Columbia (Forest and Range Evaluation Program; www.for.gov.bc.ca/hfp/frep/index.htm). Models of soil productivity and biodiversity impacts used for LCA in British Columbia will need to address the diversity of ecosystem conditions and land management practices that exist within the province and that are embedded in monitoring and evaluation frameworks through British Columbia's biogeoclimatic ecosystem classification system (Province of British Columbia 1994). Further consideration to the cumulative effects of all forms of land management that contribute significant inputs to the processes or products under investigation (Nolan et al. 2009) is also advisable. Where multiple land uses have occurred on a particular site, the manner in which biodiversity impacts should be allocated to each land use is not necessarily clear (Lindeijer et al. 2002).

British Columbia's forest land management legislation and policy framework and evaluation procedures (e.g., Curran et al. 2009; Province of British Columbia 2009) provide a starting point for sustainability evaluations within LCA. Initially, results from these non-LCA procedures could be factored into LCAs at the interpretation phase, but soil conservation and biodiversity provisions and resource monitoring programs do not yet provide the type of information needed for a life cycle sustainability assessment for bioenergy or other new bioproducts. Additional research,[7] data, and analysis would be required to obtain information on soil and biodiversity impacts from resource extraction from forests so that comparisons can be made and the results can be used in conjunction with LCA. Ultimately, the specific approach used will depend on the LCA goals and scope, types of land management practices used in the production of bioenergy feedstock, and the availability of inventory and knowledge to support indicator characterization.

The bioeconomy is expanding worldwide and British Columbia's forest lands offer significant opportunities for environmentally sustainable feedstock production. Life cycle assessment is increasingly viewed as the tool of choice for evaluating environmental, economic, and social aspects of the bioeconomy. Life cycle assessment provides a transparent, credible, and quantitative format for documenting impacts across the entire life cycle of a product, identifies where the impacts are the greatest in the supply chain, and allows rigorous comparisons to be made amongst alternatives for providing the products and services needed by society. Use of LCA may help open markets to British Columbia's new bioeconomy.

However, as desirable as it might be for British Columbia to gather and apply the data needed for LCA to support the bioeconomy, incorporating forest management impacts will be a challenge. It will perhaps require combining LCA with processes such as multi-criteria analyses and environmental performance indicators (e.g., Hermann et al. 2007), taking forest management impacts into account through parallel and complementary processes such as environmental impact assessment (Gaudreault et al. 2015; Schweinle et al. 2015), or addressing them through regulation and guidelines or third-party sustainable forest management certification instead of LCA.

RECOMMENDATIONS

1. As a starting point for LCA in British Columbia's forests, it would be prudent to initiate a project to evaluate the use of the ISO-LCA framework, together with the TRACI 2.1 tool and existing inventory data, in the bioenergy sector.

7 Specific research and monitoring approaches may need to be developed to determine, for instance, the amount and quality of forest biomass that is required to be left on site without undermining the sustainability or integrity of the system (Venier et al. 2012).

Resource specialists, scientists, and forestry practitioners would be engaged to evaluate the extent to which British Columbia's existing monitoring protocols and practices expertise could contribute information to the LCA.

2. While initial land-use impacts on soil productivity can be assessed within ISO-LCA using loss of SOM as the indicator, further work would be needed to refine an approach that would be successful in British Columbia.
3. Land-use impacts on biodiversity can initially be assessed outside ISO-LCA with British Columbia–relevant data and existing provincial assessment and monitoring frameworks; however, it might be necessary to either adapt international proposals or develop a made-in-British Columbia approach to incorporate biodiversity impacts into LCA.
4. Progress has been made in recent years in developing the infrastructure and resources to effectively apply LCA for improving policy and maintaining markets for forest products in British Columbia, but further work would be needed to:
   - encourage life cycle thinking in policy development and investment planning in the bioeconomy,
   - develop professional expertise in LCA to support industry and decision makers, and
   - develop an improved and representative Life Cycle Inventory database for British Columbia by adding to the Canadian database with ecosystem- and manufacturing-specific data.

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