Towards improved life cycle assessment of environmental systems: geographical perspectives on multi-scale challenges in bioenergy feedstock production

By

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Abstract

There is increasing interest in Ontario and other regions of Canada in expanding the use of bioenergy as part of future renewable energy production. Critical to making informed decisions on how to best support and utilize emerging bioenergy technologies depends on being able to evaluate their short and long-term environmental impacts, particularly at the biomass production stage. Accurate environmental assessments can help inform landowners, policy-makers, and energy consumers about the risks or benefits of using biomass production for energy over other land-use options such as other renewable energy projects or food production. Life cycle assessment (LCA), a tool often used to assess the potential environmental impacts of bioenergy systems, faces ongoing issues associated with evaluating dynamic multi-scale systems including environmental systems such as biomass feedstock production. In particular, the common use of large-scale averaged inventory data for a select range of conditions originated to evaluate more controlled industrial spaces; this approach is no longer adequate to evaluate certain parts of environmental systems as fine-scale characteristics of the system are highly variable, such as climate, soil, crop choices, and land management approaches. Although the importance of considering site-specific variation in LCA as geographies change has been identified for some time, adoption of geographical concepts in LCA has been slow to develop. The aim of this thesis is to facilitate a discussion between geographic and LCA thought, and address what LCA approaches should look like when we need to evaluate environmental systems across a range of geographies, in the context of bioenergy feedstock production. An in-depth analysis of existing LCA standards reveal several areas in goal and scope directions where vague or absent instructions may lead to inappropriate use of geographical attributes by practitioners. Using gaps identified in LCA standards, a systematic review of bioenergy LCA literature is performed over a ten-year period (2005 – 2015) to understand how spatial and temporal attributes of systems, input data, and scope definitions vary and may affect results when different LCAs are compared. Finally, using a modular multi-scale approach, a LCA of short-rotation coppice (SRC) willow is performed for a site in Guelph, Ontario, to evaluate modified LCA performance that incorporates geographic inputs over more traditional approaches to compiling inventories and evaluating impacts. Results show that deployment of SRC willow over a 21-year period would have net greenhouse gas and energy benefits. The inclusion of smaller scale spatial and temporal attributes in the modular approach showed that improving data resolution can be used to address site-specific questions that more aggregated data cannot. This thesis points towards a need to address issues of scale within LCA standards, particularly with respect to components of the scope such as data quality requirements. Improving the transparency of geographical attributes of future LCAs provide an opportunity to document underlying characteristics of systems that may otherwise go unnoticed, but also provide a framework in which future dynamic LCA tools and models can be integrated.

Co-authorship

I hereby declare that I am the lead author on all the chapters contained in this thesis.

Chapter 2 was developed as a review article and will be submitted for publication. Warren Mabee will be a co-author, providing input into structure and formatting as well as helping to select critical thoughts from the chapter for publication.

An early version of a working journal article that eventually evolved into Chapter 3 was commented on by Goretty Dias (University of Waterloo). Although the original paper never came to fruition, her comments provided important insight which led to decisions to completely re-evaluate the original approach taken in that paper and from which Chapter 3 evolved. The current version of Chapter 3 was completed and written independently. A forthcoming manuscript will be co-authored by Warren Mabee.

Chapter 4 uses some baseline inventory data for farm fuel and material use associated with willow biomass production in Guelph that were chosen in cooperation with researchers (Kumudinie Kariyapperuma, Goretty Dias, Claudia Wagner-Riddle, and Warren Mabee) during the production of a 2012 collaborative report for OMAFRA. A forthcoming manuscript will be co-authored by Warren Mabee.

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Effective sustainability depends on the strengths and abilities of all the components and stakeholders within a landscape or system, which allows them to contribute to each others' success cooperatively and constructively. The completion of this PhD thesis is an example of that integrative approach and could not have been possible without the support and abilities of other people, organizations, and institutions that compose my own sustainable network within my own sphere of space and time.

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Statement of Originality

I hereby certify that all of the work described within this thesis is the original work of the author. Any published (or unpublished) ideas and/or techniques from the work of others are fully acknowledged in accordance with the standard referencing practices.

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List of Abbreviations

AP	Acidification Potential
EP	Eutrophication Potential
FFU	Fossil Fuel Use
FU	Functional Unit
GHG	Greenhouse Gas
GJ	Gigajoule
GWP	Global Warming Potential
ha	Hectare
iLUC	Indirect Land Use Change
IPCC	International Panel on Climate Change
ISO	International Standards Organization
LCA	Life-cycle Assessment
LCI	Life-cycle Inventory
LCIA	Life-cycle Impact Assessment
LUC	Land Use Change
MJ	Megajoule
MWh	Megawatt Hour
PJ	Picojoule
odt	Oven-dried tonne
SRC	Short-rotation coppice
SOC	Soil Organic Carbon
USDA	U.S. Department of Agriculture

Chapter 1 – Introduction

1.1 Background

1.1.1 Bioenergy Trends

The finite reality of fossil fuel resources and the climate change effects associated with their use have incentivized the pursuit of alternative and renewable sources of energy. While some renewable sources, such as wind and solar, have proven to be viable options for electricity production, they are currently limited in their ability to effectively supply energy for sectors such as road transportation and heating, and justify the need to explore other alternatives such as bioenergy and biofuels (Giuntoli 2013; Ridley et al. 2012). Functional advantages of bioenergy over some other renewable sources include its capacity to deliver large amounts of thermal energy that can be efficiently used for heating (Keoleian & Volk 2005; NRCan 2016a) and its ability to be integrated in biofuel form into existing transport fuel infrastructures (Lönnqvist et al. 2015; Simone 2009). It also has the ability to improve energy security by reducing energy supply uncertainty and price fluctuations (Keolean & Volk, 2005), and the ability to be stored long-term and used on demand, which can help address intermittent supply issues associated with some other renewable sources (Bauen 2009). On a global scale, demand for energy is expected to rise 37% by 2040 (IEA 2014); it is estimated that the amount of energy produced from bioenergy could triple by 2030 to help meet this demand (IRENA 2014).

In Canada, energy produced from renewable sources contributed 1989 Petajoules (PJ) in 2013, or 11% of the total national annual production (NRCan 2016b). While most of the energy produced from renewables was from hydro (70.7%), bioenergy contributed nearly all of the rest, composing of solid biomass to heat and electricity (24%), ethanol in gasoline (1.8%), and biodiesel in diesel fuels (0.2%). Energy produced from wood waste streams (liquid and solid) has remained relatively consistent (380 PJ/year) from 2000 to 2013. The NRCan (2016b) report also stated that energy from pellets as a solid fuel has increased in its contribution to total energy produced; pellets combined with fuelwood contribute approximately 200 PJ/year to Canadian energy supply.

Industrial (59.4%) and residential (31.4%) bioenergy applications dominate, with a smaller portion diverted to electricity production (9.2%) (NRCan 2016b). Approximately 10% of total energy used by industry comes from wood waste and spent pulping liquor, which highlights the integration of bioenergy within the forest sector in particular. Atikoken, a former coal-fired power plant (900 MW capacity) that came online in 2014, is currently being fueled exclusively using forest residue feedstocks (Bradburn, 2014). Residential use of bioenergy is dominated by demand for heat; 19% of all residential heating needs are met through bioenergy (Cuddihy et al. 2005).

When energy production is compared across different geographic areas of Canada, renewable or otherwise, there are distinct trends that emerge. For example, annual energy production in western provinces (2015 data) is dominated by crude oil (>7000 PJ), natural gas (>5500 PJ), and coal (>1000 PJ). Hydropower is predominantly generated in Quebec, Ontario, Manitoba and British Columbia, where both the availability of fresh water and the space and landscapes available to execute such projects are present. Other renewables (~1000 PJ), which are primarily composed of wood and wood waste energy streams, are more likely to be produced in forestry-oriented provinces like Quebec, Ontario, and British Columbia (NRCan 2016b).

In Ontario, there is increasing interest in the potential for existing or unused agricultural lands to be used to cultivate dedicated energy crops, such as those derived from agroforestry and agricultural feedstocks (Zegada-Lizarazu & Monti 2011; Zhu & Zhuang 2012; Dhillon and von Wuehlisch 2013). Energy crops from these types of systems have been explored as a viable source of biomass energy, particularly in North America (Volk et al. 2006; Keoleian & Volk 2005) and Europe (González-García et al. 2013; Murphy et al. 2014). Results from a number of studies have endorsed their deployment as part of a larger renewable energy approach. While first generation biomass crops focused on the use of annual cellulosic crops including residues from cereal crop production, there is increasing interest in the use of perennial lignocellulosic crops such as switchgrass (Murphy & Kendall 2013), willow, poplar, and eucalyptus (Pleguezuelo et al. 2014).

1.1.2 Willow (*Salix* spp.) as a Bioenergy Feedstock

One feedstock gaining interest as a potential bioenergy source in Ontario is short-rotation coppice (SRC) willow (*Salix* spp.). Modern interest in the use of willow for energy in Canada largely originated out of the 1970's oil crisis and the subsequent creation of ENFOR (Energy from Forest), a federal program targeting energy research and development for the forest bioenergy sector (Kuzovkina et al. 2008). In addition to a renewable source of energy for heat, power, and transport biofuels, when willow coppice biomass is included as part of a bioconversion platform, it has the potential to provide important value-added bioproducts from biorefinery scenarios (Guo et al. 2016; Krzyżaniak et al. 2014).

1.1.2.1 Willow Cultivation

Although the 330-500 species of *Salix* spp. are largely found in temperate and arctic regions, there are some subtropical and tropical species as well (Karp et al. 2011). Biodiversity of willow is greatest in Asia where there are approximately 275 species (Kuzovkina et al. 2008); North America is home to 103 different species (Argus 1997). While the genus of *Salix* can be divided into four sub-genera (*Salix*, *Longifoliae* Anderson, *Vetrix* Dum and *Chamaetia* Nasarow), most willow species currently in use by commercial plantations belong to the subgenus *Vetrix* and mainly take the form of small trees and shrub willow (Kuzovkina et al. 2008; Karp et al. 2011).

A number of biological and ecological features of *Salix* species have contributed to its selection for cultivation. For example, *Salix* can reach maximum annual growth in only a few years following establishment, and can tolerate high planting densities improving its yield per area of production and ability to be easily adapted into existing agriculture systems (Keolein and Volk 2005; Karp et al. 2011). The fast growth of willow is partly due to the ability of *Salix* for high light-use efficiency when water and nutrients are not limited (Tharakan et al. 2008; Cannell et al. 1987). Species of *Salix* also have the ability to coppice (or re-sprout) after harvest, which permits repeated and frequent harvesting at low cost relative to traditional forestry operations (Keolein and Volk 2005). Typically, woody coppice systems such as willow are harvested every 3-5 years, and are believed to remain viable for 15-30 years before the crop should be removed and new cuttings will need to be re-planted (Aylott et al. 2008).

Many *Salix* species utilize strategies that take advantage of newly opened habitats where they are often early successional plants (Kuzovkina et al. 2008). Early succession adaptations allow *Salix* to colonize habitats that have been influenced by a number of forest disturbances. The species has the ability to be grown on marginal or degraded agricultural land and under conditions that would otherwise be unsuitable for conventional crops, reducing the need to compete with food crops as a bioenergy feedstock (Zamora et al. 2014; McElroy & Dawson 1986; Amichev et al. 2012). Willow also has typically longer growing seasons than *Populus* species (Cannell et al. 1988).

A number of field trials and breeding programs for *Salix* have been initiated over the last few decades, and a number of them have focused on selecting certain cultivars that show faster growth traits (Smart & Cameron 2008; Gullberg 1993). Others have focused on other physiological differences observed such as the tendency for some varieties to partition greater fractions of biomass to harvestable stem portions (Karp et al. 2011). Although many field-trials and breeding programs have focused on carbon balances and improved yields through phenological selection and management strategies (Liu 2013), some studies have also considered capacity for erosion control (Wilkinson 1999), effects on biodiversity (Dhondt et al. 2004; Nerlich et al. 2012; Nakamura et al. 2006), and potential for soil remediation (Aronsson & Perttu 2011).

1.1.2.2 Environmental Impacts of Willow Plantations

The integration of bioenergy crops, such as SRC willow, into existing or previously used agricultural landscapes means that their production and their associated land management activities may interact with and impact the environment differently than crop agroecosystems or traditional woodland landscapes. These impacts may be beneficial; willow biomass crops can positively influence regional and global carbon cycles by absorbing carbon through photosynthesis and sequestering it in above ground and below ground biomass (Volk et al. 2004). The ability of *Salix* to produce significant root biomass is of particular interest for carbon sequestration and soil health (Rytter & Hansson 1996). How SRC willow growth is influenced by, and interacts with, the surrounding environment has only been studied long-term in a few locations (Liu 2013; Volk et al. 2011). Even though stool (stump) and root biomass health can be critical for improving willow crop growth – particularly where their development can be directly correlated to aboveground yields (Rytter & Rytter 1998) – there is still relatively little known about the belowground portion of willow dynamics (Volk 2001). Due to the relationship between crop residue deposition and soil carbon sequestration, gaps in understanding willow growth dynamics also means that the effect of SRC willow on soil carbon as the plantation site changes is not well known.

Variability between sites, such as edaphic conditions and local land management approaches, can also influence feedstock yields (Djomo et al. 2011). For example, SRC willow yields observed following the first rotation ranged from 0.3-27.5 Mg/ha/yr across a number of studies reviewed in North America and Europe (Liu, 2013). Willow yields have been shown to increase under management regimes that add nitrogen-based soil amendments such as slurry (Labrecque & Teodorescu, 2003) or chemical fertilizers (Heinsoo et al. 2009). It is possible to observe differences in willow yields at the same location at the farm-scale. In work performed in Guelph, Ontario, willow yields following the first rotation ranged from 6.72 – 13.5 odt/ha/yr, depending on the variety of shrub willow grown (Clinch et al. 2009). There is some evidence that breeding programs over the last few decades have improved yields across a number of early SRC willow species (Liu 2013).

Highly variable willow yields - influenced by site-specific environmental, crop, and land management details - ultimately contributes to the range in potential energy production from SRC willow plantations reported on a per area basis. For example, Heller et al. (2003) reported a bioenergy to fossil energy input ratio for SRC willow of 55:1 while Roedl (2010) found the value to be 43:1; both values are within the ranges found by Matthews (2001) (20:1 – 64:1), but nearly twice the value (22-28:1) observed by Dubuisson and Sintzoff (1998) in Belgium. In order to provide land-owners, policy decision-makers, and energy consumers in Ontario with information about the long-term competitive advantages of willow over competing land-use alternatives, there is a need to understand how site variability influences the environmental impacts or benefits of bioenergy feedstocks such as SRC willow (Campbell et al. 2013; Cherubini & Strømman 2011; Gold & Seuring 2011). Competing land-use alternatives can include other agriculture crops, but may also include other competing renewable energy options (Calvert & Mabee 2015). To summarize, energy production from biomass feedstocks such as willow is difficult to justify without a better understanding of potential land-use impacts.

1.1.3 Life Cycle Assessment (LCA) Challenges

Currently, policy decisions can be aided by a standardized method of assessing the environmental impact of industrial and power generation projects, called Life Cycle Assessment (LCA). This methodology was designed to consider all aspects related to the production, use, and ultimate disposal of a product, including inputs and outputs from each stage in the value chain (ISO 14040, 2006; ISO 14044, 2006). LCA can be used to evaluate the impacts of specific products (like bioenergy crops), but the metrics used to within an LCA are not standardized; they can include a broad range of environmental and socioeconomic inputs (energy, labour, chemicals, etc.). Evaluations provided by LCA are usually expressed through midpoint analyses that provide a measure of global warming potential (i.e. greenhouse gas emissions), acidification, or eutrophication, or endpoint analyses which provide a combined measure of total impacts.

LCA methodologies are challenged when they are used to assess product system boundaries that might include processes from open and dynamic environmental systems, which include biomass generation in both forestry and agricultural sectors. It has been observed that agroecosystem LCAs on similar systems can generate conflicting results, as the dynamics of the specific system being assessed can deviate significantly from the mean input values used to define those systems. Thus, the common LCA approach – using average inventory values or emission factors to quantify potential environmental impacts – can become increasingly inappropriate as geographies change and the environmental characteristics under which data being used were obtained deviate further from the current system being assessed (Collet et al. 2014; Godard et al. 2013; O'Shea et al. 2013).

To quantify the impacts of emerging products and services that are derived from environmental systems, there is a need for LCA methods to address the variability associated with spatial and temporal scales – to consider the impact of geographies and different growing seasons on the overall impacts of the product. For emerging bioenergy technologies, which depend on feedstocks produced exclusively from environmental systems, overcoming methodological challenges is paramount to ensuring that climate change and sustainability goals are being met. Biomass production systems that change across geographies or vary year by year depending upon climate may require a variety of scales of analysis, in order to apply LCA appropriately as system-specific characteristics change (Reap et al. 2008; Sanden & Karlstrom 2007).

Often differences in LCA results between bioenergy studies have been attributed to the different methodological choices needed for data collection and interpretation (Thomassen, 2008); the range in methodological approaches used from one location to another has produced equally variable results and led to conflicting conclusions (Whitaker et al. 2010; Weidema 2014). Traditionally, LCAs apply average inventory values or emission factors from large aggregated datasets in order to account for site-specific processes. The dynamic behavior of ecological systems implies that the traditional steady-state, closed system approach to compiling life cycle inventories creates a barrier to updating changes that may result from new site or study-specific characteristics (Reap et al. 2008; Sandén & Karlström 2007). For a country such as Canada – even in a province such as Ontario – regional climate and landscapes can change dramatically from place to place or year to year, meaning that models that ignore site-specific processes may generate decreasingly reliable estimations as land-use and context change. To provide agricultural producers of best-practice scenarios for sustainable bioenergy crop production with better information, the LCA framework must reflect changing management options and practices that may change with space and time.

1.2 Research Questions & Objectives

While the importance of considering geographical variability in LCA has been identified, actual adoption of geographical concepts in LCA and movement towards addressing issues of scale has been slow to develop. In order to facilitate a discussion between geographic and LCA thought, the overarching question addressed in this dissertation is what should LCA approaches look like when evaluating environmental systems whose characteristics can change across geographies? The specific research questions asked in this thesis, which aim to address the central question in more detail, are as follows:

1. To what degree do LCA standards currently direct practitioners on how to address geographical attributes of data used?

2. What are some of the characteristics observed in agroecosystem dynamics that are important, from a geographical perspective, to consider in life cycle impact assessments?

3. How reliable are comparisons between different LCA results when we take into consideration the spatial and temporal attributes of underlying assumptions and data quality?

4. Does the scale of the product supply chain influence LCA data choices more or differently than the spatial and temporal resolution of data available?

5. Can a modular approach to multi-scale LCA systems facilitate better interpretation of LCA results when there are a number of data options available?

6. What should the LCA framework encompass moving forward to better integrate site-specific concerns and address evolving land-use questions.

It is hypothesized that gaps in the way that LCA standards direct practitioners on how to handle geographical concerns cause significant difficulty in terms of interpreting results between specific sites. Incorporating geographic thinking in order to address issues of spatial and temporal scale it could provide LCA with a better perspective on how to evaluate environmental impacts of multi-scale systems. To answer the primary research questions of this thesis, three complimentary objectives are defined:

1. To identify ways in which geographical concepts can be best integrated into life cycle assessment, using bioenergy systems as an example.

2. To assess the ways in which geographical concepts of scale have been addressed in existing bioenergy LCAs, and observe how they may influence life cycle results.

3. To explore the use of modular approaches to LCA as a means of solving issues of scale and dynamism; a case study of willow short-rotation woody crop systems in southwestern Ontario is analyzed for this work.

1.3 Discussion of Research Objectives and Chapter Organization

A number of reviews and articles that have discussed the differences between various LCA components. These include allocation procedures (Ekvall & Finnveden 2001; Schmidt 2008; Luo et al. 2009; Kaufman et al. 2010), consequential and attributional approaches (Sandén &

Karlström 2007; Zamagni et al. 2012; Marvuglia et al. 2013), perennial crop variations (Bessou et al. 2013), and system boundaries choices (Ekvall & Weidema 2004; Suh et al. 2004; Roer et al. 2012). None of these studies have yet explored how underlying spatial and temporal attributes of LCA assumptions and methods might relate to environmental performance, or considered the transferability of methods over both space and time. The main objective of Chapter 2 is thus to help close this gap in the form of a critical review of the current LCA standards. The chapter highlights how geography is addressed in current LCA standards, and identifies existing limitations in LCA standards where geographical concepts can and should be considered in order to improve our understanding of environmental impacts as spatial and temporal characteristics of systems being assessed change.

In Chapter 3, a comprehensive literature review of bioenergy LCAs over a ten-year period (2005-2015) is performed, with an emphasis on summarizing how the spatial and temporal attributes of system characteristics such as boundary definitions, system process flows, and methodological assumptions are handled in each article and thus can effect overall results or conclusions. The results and biomass feedstock outputs are compared with the geographical characteristics implied by choices made in each study. The central hypothesis of Chapter 3 is that while LCAs may seem comparable on the surface because of similar feedstocks, functional units, or supply chain scopes, in reality they cannot be easily compared due to variations in the underlying spatial and temporal data and system definitions used. It is expected that the spatial and temporal differences in the way that data inputs and system boundaries are scaled between LCA components limits the ability for results to be compared between studies, and limits their value as the geography of the assessment changes.

In order to address the impacts of sustainable energy program delivery in Ontario, resource managers need the ability to assess where the biggest environmental impacts are within the life cycle of bioenergy crops. Chapter 4 uses the case study of willow production in southwestern Ontario to explore this issue. The main research problem is how to appropriately account for the environmental impacts of biofuel feedstocks within biofuel production; while agricultural and agroforestry systems have the potential for significant environmental impacts, there are also potential benefits to air, water, soils, biodiversity and food production which will require spatially and temporally specific application of LCA. Therefore, in order for a LCA framework dedicated to improving bioenergy production performance to be successful, and to inform agricultural producers of best-practice scenarios, it must be adaptable to different environments, changing management practices, and a variety of agricultural technologies. It is important that future agroecosystem LCAs consider dynamic activities and impacts at the farm or field scale, but it is equally important that they provide feedback to agricultural stakeholders, in order to improve resource and land management sustainability (Horne et al. 2009).

Chapter 4 evaluates a modular dynamic LCA framework that can integrate processes and activities from a range of spatial and temporal scales. This approach allows for simultaneous evaluation of impacts and technological change across a range of data with varying quality, which has been an ongoing issue with traditional static systems and emission factor approaches of traditional LCAs. This approach has the potential to improve feedback information to agricultural stakeholders, and enhance their response to future agroecosystem due to its input and output adaptability. The ultimate end goal is to be able to help inform sustainable energy policy in Ontario that can make the most efficient use of available resources within a landscape, because resource management needs the ability to assess relative environmental and social impacts across the life cycle of emerging energy technologies. If successful, this assessment has the potential to provide a framework for policy makers to develop green energy policies that will maximize efficiency of renewable energy sources, while simultaneously informing agricultural stakeholders of appropriate approaches to improve the sustainability of biomass feedstock production.

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Chapter 2 – Critical exploration of life cycle assessment (LCA) standards through a geographical lens

Abstract

The practice of life cycle assessment (LCA) is challenged when confronted by open and dynamic multi-scale systems; this can make evaluating emerging bioenergy technologies difficult as biomass feedstocks are often produced across highly variable landscapes. The effects of local environmental conditions and land management activities have been slow to be integrated into LCA. This highlights a need to improve geographically explicit approaches to LCA, and to improve methodological instructions for dealing with spatially and temporally explicit data within LCA. This research examines LCA standards through a geographical lens to understand how geography attributes of system boundaries and underlying unit process are addressed within the standards, and explores the implications of these findings when LCA is applied to bioenergy systems. Findings indicate that several components of the LCA scope are inherently linked to geographical attributes; however, direction on how to manage geographical data and in particular spatial and temporal scales are lacking, particularly in components defined within the goal and scope of the study. Areas within the standard that could be improved by adopting more geographically explicit language and criteria are identified, and pathways for more forward-thinking LCAs are discussed.

2.1 Introduction

While some studies, including those carried out in Canada and in Ontario, have proposed that net reductions in greenhouse gas (GHG) emissions are possible when biofuels are used in place of petroleum fuels (Pelletier et al. 2008; Whitman et al. 2011; Spatari et al. 2005), results between studies can vary significantly. For example, while some studies such as Kim and Dale (2005;2008) and Shapouri et al. (2002) have found that corn-based ethanol produces less GHG emissions compared to fossil fuels, their results are disputed by other studies such as Pimentel et al. (2009). Also of concern are a number of articles that suggest that when impacts besides global warming are considered, such as eutrophication and water degradation, they may negate or even reverse the environmental benefits of sequestering carbon through biomass growth, (Yang et al. 2012; De Fraiture et al. 2008), suggesting that a more robust set of environmental criteria should be considered. Others have argued that ignoring direct land use change (LUC) impacts such as the loss of biodiversity or soil carbon may misrepresent the benefits of agricultural expansion (Searchinger et al. 2015). Bioenergy has also been charged with causing significant indirect LUC impacts, such as increased deforestation due to expansion of agricultural land (Tsao et al. 2012; Hertel et al. 2010; Hosonuma et al. 2012), and has been linked to increasing demand and prices for agriculture commodities (Johansson et al. 2014; Baral & Malius 2016; Searchinger 2011). The result has been little consensus on the sustainability of bioenergy and has led some to question the suitability of LCA under its current structure to address these issues.

Part of the challenge when LCA is used to evaluate bioenergy technologies is that unlike the more controlled indoor spaces of traditional industries, which dominate much of the early LCA literature, biomass feedstock production exists in open systems that are connected to highly variable background environments (Haas et al. 2000). Site-specific geographical changes at the farm-scale can be highly variable and directly affect environmental performance. Of particular interest are specific changes in local climate and weather patterns (Ericsson et al. 2013; Kucharik & Ramankutty 2005), varying agricultural practices (Rothenberger et al. 2009; Salmon-Monviola et al. 2012; Uchida & Hiyashi 2012), and the cost and availability of technology (Ciroth et al. 2002; Pehnt 2006; Svoboda et al. 2009). The environmental performance of bioenergy is dependent on complex interactions between human activities and environmental processes where variability in local factors can directly affect the growth of biomass crops (Hessburg et al. 2005), and by extension influence how crops are chosen and managed (Fazio & Monti, 2011). This is why it is possible to have conflicting study results published describing similar feedstock systems from comparable temporal periods.

The influence that site-specific characteristics can have on life cycle assessment (LCA) results has been identified as a significant challenge for some time and led to calls for more geographically explicit approaches (Gold and Seuring 2011; Reap et al. 2008; Singh et al. 2010). Despite this, the progress of LCA methodologies towards addressing integrated human and environmental processes across space and time has been slow. There is a noted void in LCA methodologies that might direct practitioners towards appropriate methods to deal with geographically linked changes, resulting in numerous interpretations of its application (Ruviaro et al. 2012; Weidema 2014). Without direction on how to manage geographical attributes of data or subsystem components within LCA standards, it is important to observe that it remains unclear as to the most appropriate way to consider geographical influences when performing LCAs. Given current challenges faced by LCA, there is potential opportunity for geography to facilitate insight and bridge conversations. As a discipline, geography offers a potential lens to manage temporal and spatial issues in a coordinated way; geography is not solely about identifying where objects exist on a map, but how the characteristics and patterns of those objects have changed and manifested, which often includes an evaluation of objects through time as well as space.

This article reviews current ISO standards (ISO 14040:2006 and ISO 14044:2006) and describes the extent to which current LCA standards, in the context of bioenergy assessments, direct practitioners on how to address geographical attributes of product systems, unit processes, inventories, and interpretation of LCA results. To date, a few attempts at improving the integration of geographical attributes in LCA have focused on improving spatial characterization within life cycle inventories (Suh & Huppes 2005; Koellner et al. 2012). It has also been shown that temporal attributes can vary in the functional unit (FU) and life cycle impact assessment (LCIA) (Collet et al. 2014); this suggests that geographical challenges may influence multiple components of the LCA framework, requiring a more thorough evaluation of theses challenges within LCA standards. Specifically, this review examines components defined in early goal and scope stages that have important methodological influences on subsequent LCA stages, and how limits placed on their definitions or absence in how to define those components can affect the application of LCA when assessing agroecosystems at different scales. Geographically explicit challenges may manifest differently depending on which aspect of the LCA framework is being considered and implies a need to look at some of these components independently and in-depth, in order to explore where spatial or temporal issues of scale and geography apply.

2.2 Reviewing LCA standards – Highlighting Geographical Concerns

The LCA framework is defined by four generalized stages of application: the goal and scope, the LCI, the LCIA, and the interpretation stage (Figure 2.1). An important point to note about the illustrated framework is the interconnected relationship that each of the four stages has with the others, emphasizing that decisions made in each stage of an LCA may require revisiting decisions made in previous ones. In this respect, of particular interest within the standards are those that guide practitioners in defining the goal & scope of their assessment. An LCA is initiated by defining the goals of the LCA, which presents the reason for the assessment and addresses its intended application and audience (ISO 14044, 2006). The scoping phase of the LCA framework is where many of the choices about assessment objectives and limits occur, and these may critically influence decisions made in subsequent assessment phases (such as inventory data choices, or how to apply impact factors) (Guinee et al. 2011). In the following sections, each of the main areas of the LCA framework are explored and components that have the potential to be linked to geographic attributes within a biomass feedstock production system are highlighted, and show links between earlier methodological decisions where appropriate.

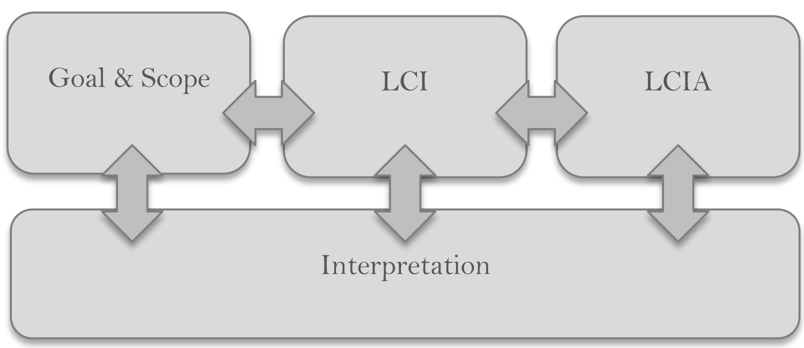

Figure 2.1. General framework of the four stages of the LCA method as set out in ISO 14040 (2006): Goal & Scope, Life Cycle Inventory (LCI), Life Cycle Impact Assessment (LCIA), and Interpretation

2.2.1 Goal & Scope Components

The scope portion of the goal and scope phase has 12 individually defined components that are part of defining the scope of the assessment (Table 2.1). The scope components explored further in this review include decisions associated with defining product systems and their boundaries, decisions about the functional unit, decisions associated with choosing appropriate data requirements, and finally the impact categories that these decisions can affect. What is important about the components identified above is that they have influence across all four stages of the LCA, but can also be related to geographical attributes of the product system and underlying unit processes. The following sections summarize relationships observed between components and discuss some of the implications they may have for bioenergy LCAs.

Table 2.1. List of items included in the scope, as set out in ISO 14044 (2006).

List of Components Included in Scope	
• The product system to be studied	• Data requirements
• Function of product system (in case of comparative studies, the systems)	• Impact categories and method of impact assessment, and interpretation
• The functional unit	• Limitations
• The system boundary	• Initial data quality requirements
• Allocation procedures	• Type of critical review, if any
• Assumptions	• Type and format of study report

2.2.2 Product Systems & Functional Units

The LCA scoping exercise is initiated by defining the product or service to be assessed and its associated function. The importance of choosing a functional unit is to ensure that data from input and output processes are normalized and therefore comparable. The functional unit also represents the conditions in which the results are shown and analyzed. It is imperative that reference units be clear, as absolute reductions in emissions reported do not allow for comparison to fossil fuels, other forms of renewable and non-renewable energy, or even other LCAs (Cherubini et al. 2009). Some typical bioenergy examples often subjected to LCA include bioethanol, whose function is to fuel a passenger car, biogas used to generate electricity, or wood pellets that are combusted for heat or power generation.

How the function of a bioenergy product is measured can vary, so the choice of a functional unit (FU) that appropriately represents the functional flow of the product system being assessed is another important component of the scope. The types of FUs chosen tend to represent end-use products (for example, all inputs and outputs might be reported on a per litre of fuel produced or per unit of energy produced basis) rather than intermediate products (for example, it is rare to see inputs and outputs reported per unit of feedstock consumed) (Choudary et al. 2014; Miller 2010). It should be noted that there are an increasing number of LCAs that are choosing to report using FUs that are more geographically-focused (for example, units per hectare). This is part of an effort to improve farm activity details that tended to get lost in many of the earlier bioenergy LCAs that focus on end-products that encompass large national or global scales of assessment (Goglio et al. 2012).

While the FU can represent product systems of varying scale, an important observation to make is that FUs can also include spatial or temporal units of measure. Sometimes FUs may only include one type of unit, either spatial (e.g. units per ha: see Spatari et al. 2005; Blengini et al. 2013) or temporal (e.g. units per year: see Eranki & Dale 2011; Gnansonou et al. 2009), while others may include both (e.g. units per ha per year: see Gonzalez-Garcia et al. 2013; Godard et al. 2013). What is apparent from literature is that while the FU may include spatial or temporal units that are abstract on their own, they may also include units that are highly specific to a particular geographical spatial unit or region; for example, Tonini et al. (2012) reported on a per ha basis but their study was specific to Danish agricultural lands.

When looking at the way that bioenergy LCAs define the FU in comparison to the product function, the scale of the relationship between the two is not always the same between studies. For example, in studies that employ a functional unit that contains a relatively small spatial unit of measure (e.g. units per ha/yr), the scale of the product system can vary from national scales (e.g. Gonzalez-Garcia et al. 2012), to regional or state scales (e.g. Eranki & Dale 2011), or even county scales (e.g. Geyer et al. 2013). There is a wide range in the difference of spatial scales between functional units and the product systems they represent. The functional flow in many ways is an expression of the resolution at which the product system is detailed and many LCAs are using this relationship, at least in terms of the units of measure they are using, to anchor the geographical resolution of the FU to the product system. For LCA practitioners that understand the geographical importance of particular aspects of their study, but lack the direction to use spatial and temporal attributes, it is common to tag the functional unit or product system with geographical information. This has important consequences. Assessments that use common FUs may conceptually allow the impacts of different products (serving similar functions) to be compared. As an example, the total MWh of electricity produced from units of coal may be compared with similar production from units of wood. When we examine how the FU relates to the geographical realities of the actual system under assessment, however, it may be observed that a common functional unit does not necessarily imply that the underlying spatial and temporal variations in the subsystems are the same. This potential making their comparisons less appropriate than existing standard definitions might suggest.

2.2.3 System Boundaries, Processes, & Data Quality

Determination of appropriate product system boundaries is guided by an assessment of which unit processes are likely to significantly impact results, and thus should be included in the assessment (Ekvall & Weidema 2004; Suh et al. 2004). Common subsystems included from upstream processes can include fertilizer production and transportation of goods, while on-farm subsystems are likely to include cultivation & maintenance, harvesting activities, and biomass growth. The variability of agroecosystem production means that depending on the objectives of the LCA or the product pathways being evaluated, which subsystems are included and the detail that each should be considered at can vary considerably from one study to another.

When practitioners look to LCA standards for guidance on choosing the appropriate level of detail needed to accurately reflect the environmental performance of system processes, there are criteria in the 'data quality requirements' component of the scope (Section 4.2.3.6.2) aimed at helping them decide whether data being used to represent product system processes is appropriate (Table 2). Criteria that directly address geographical attributes are limited though. Spatially, the standards define 'geographical coverage' as the "geographical area from which data for unit processes should be collected to satisfy the goal of the study" (ISO 14044, 2006). The LCA standards are therefore using the term 'geography' merely as a way to define a particular location or area where an assessment is being conducted, but fail to elaborate on how locations or areas of differing sizes or resolution should be identified or labeled relative to other geographic areas. While geographical information may provide direction on how to structure and manage underlying data of models, it was never considered to be a necessary component of the modeling framework in order for LCAs to be able to function, at least in the formative years of the methodology (Bengtsson et al. 1998).

Given the focus of LCAs on end-product assessments, combined with reality that feedstock production is often distributed as heterogeneous patches across a larger landscape, bioenergy system boundaries tend to include processes that exist over large geographical areas. Spatially large product systems can enable practitioners to choose emission factors or averaged values from large spatial areas when data is being collected for life cycle inventories. This means that, depending on the scale or scope of the assessment, certain processes or activities may be ignored while others may dominate. For example, although applying national emission factors or average farm data inputs may satisfy data requirements associated with geographical coverage, using data aggregated from large scales can easily lead to significant over- or –under-estimation of yields at a specific place or time. When examining agriculture census data available from the U.S. Department of Agriculture (see USDA 2014) agronomic data is predominantly aggregated to national and state levels with some county data available for select years and regions. In 2012, national corn grain yields (171 bu/acre) were comparable to state averages in Nebraska (170) and Texas (174), but much lower (around 153 bu/acre) than state yields from the Midwest (Indiana, Iowa, and Missouri). Yields tend to vary even more at smaller county level averages where county production ranged from 80 – 135 bu/acre in Iowa while neighbouring Nebraska yields were 46 – 185 bu/acre. Aggregated data are thus likely to be inappropriate at all locations within a geographical area as the spatial resolution of the system changes and may lose some of the critical small-scale geographical information that might be important to keep track of at more local, geographically explicit levels of detail.

Variability in agronomic data at different scales of aggregation is not limited to spatial phenomenon. For example, national U.S. corn grain yield averages were reduced by 9 bu/acre between 2007 – 2012, while state yields in the Midwest region dropped between 24 bu/acre (Nebraska) and 70 bu/acre (Illinois). Variation at the county level differed even more within states where Missouri yields saw a +14 to -85 bu/acre change. Although temporally aggregated data can mask time-sensitive variations, there is a similar lack of direction in data quality requirements associated with temporal data where standards direct users towards ensuring 'time-related coverage' that is defined as the "age of data" and the "minimum length of time from which data should be collected" (ISO 14044, 2006). Since yield measurements usually require a full growing season, appropriation of yields from a single year can be justifiable given the lack of focus on short-term or long-term variability.

Table 2.2. List of the data quality requirements and their associated descriptions as set out in ISO 14044 (2006), section 4.2.3.6.2.

Requirements	Description
a) time-related coverage	• Age of data and the minimum length of time over which data should be collected
b) geographical coverage	• Geographical area from which data for unit processes should be collected to satisfy the goal of the study
c) technology coverage	• Specific technology or technology mix
d) precision	• Measure of the variability of the data values for each data expressed (e.g. variance)
e) completeness	• Percentage of flow that is measured or estimated
f) representativeness	• Qualitative assessment of the degree to which the data set reflects the true population of interest (i.e. geographical coverage, time period and technology coverage)
g) consistency	• Qualitative assessment of whether the study methodology is applied uniformly to the various components of the analysis
h) reproducibility	• Qualitative assessment of the extent to which information about the methodology and data values would allow an independent practitioner to reproduce the results reported in the study
i) sources of the data	• (none provided)
j) uncertainty of the information	• (e.g. data, models and assumptions

2.3 Life-cycle Inventory (LCI) Analysis

Life-cycle inventory (LCI) analysis is required to quantify the inputs and outputs for each unit process included as part of the system boundary. Reliance on system boundary definitions for guidance when performing LCI analysis highlight the two-way co-operative influence that the goal an scope should have with the inventory where definitions can be refined, as shown earlier in Figure 1. The ISO standards currently provide only general direction on choosing appropriate boundary cut-off points by emphasizing that i) flows in and out of the system should not be transformed or manipulated upstream or downstream of the system, and ii) if they are then the system should be expanded to include them until no more transformations are observed (ISO 14044, 2006). LCI analysis procedures therefore provide *some* additional criteria that practitioners can use to determine appropriate levels of data to quantify inputs and outputs of unit processes.

There are three general procedures identified in the LCI phase: data collection, data calculation, and allocation. As the LCI proceeds through these procedures, there is a series of operational steps outlined to guide data collection and its subsequent application to unit process calculations, and includes direction on when to consider revisiting system boundary definitions (Figure 2). During collection procedures, data can be classified under a number of headings depending on system-specific operating conditions associated with each unit process; in the context of bioenergy supply chains where feedstock production is included these can include but are not limited to energy inputs, raw material inputs, or releases to the environment.

Data calculation procedures encompass several of the operational steps illustrated in Figure 2.2, which can be broadly categorized as data validation, relation of data to unit processes and functional unit, and finally system boundary refinement should it be required. An important part to identify in the validation stage is the continued dependence on criteria defined during the goal and scope phase where data is compared to data quality requirements. As identified previously, while the criteria in the requirements may guide geographic coverage they do not discuss the relation of that geographic extent to spatial or temporal resolution of the underlying unit processes. Similarly, relating data to the unit process and functional unit requires referencing all input and output data to the functional unit, also previously defined in the goal and scope. As such, shortcomings in how functional units are defined - previously due to vague geographical direction - have the potential to be duplicated in the LCI analysis.

It is important to note that standards defined under Section 4.3.3.3 of ISO 14044 standards do address the careful consideration needed when aggregating inputs and outputs within the product system. They remain vague in stating that, "If more detailed aggregation rules are required, they should be explained in the goal and scope definitions phase of the study or should be left to a subsequent impact assessment phase." (ISO 14044, 2006; pg 13). The open nature of bioenergy production systems makes interpreting and applying these standards to environmental systems, such as crop production, less straightforward when compared to industrial production systems. LCA standards are unclear on how to address and manage underlying spatial and temporal attributes; however, the traditional use of mean values aggregated from a number of sources may lose some critical geographical information – either spatial or temporal (Cooper et al. 2012; Finnveden et al. 2009; Rebitzer et al. 2004). One example where aggregated data might not be able to reflect the influence of spatial variability is in relation to fugitive emissions. There are many fugitive emissions that are difficult to measure, such as GHG emissions or soil leachates, which vary spatially at the field-scale depending on soil heterogeneity (van Groenigen et al. 2011).

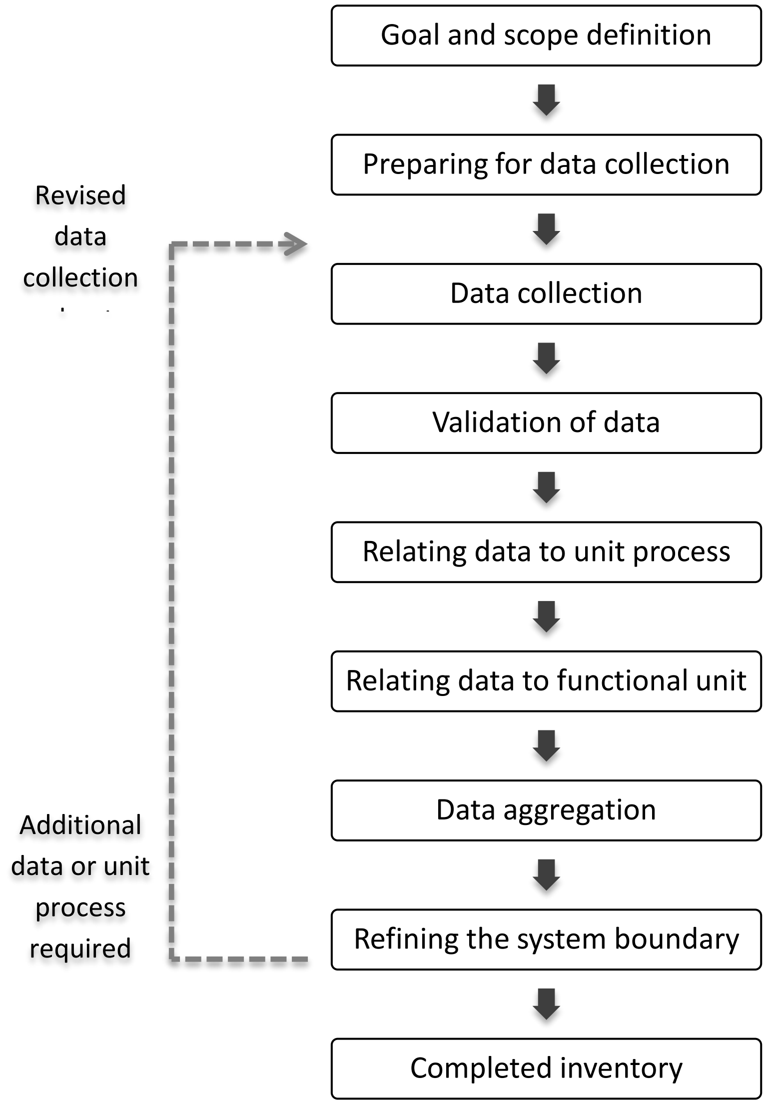

Figure 2.2. Flow chart illustrating an adapted version of the 'simplified procedures for inventory analysis' figure presented in the LCA standards (ISO 14044, 2006; Figure 1, pg 12).

The only other device described in the LCI phase that guides practitioners in determining the level of detail required when compiling inventory data is the sensitivity analysis, as set out under the 'Refining the system boundary' heading (section 4.3.3.4). A sensitivity analysis is not useful if some of the details of the data are missing. Unless the inventory data incorporates fine-scale changes such as soil carbon responses to residue harvesting or crop cover types, it is difficult to see how the sensitivity analysis will capture these issues. While a sensitivity analysis under current definitions may result in exclusion of data or inclusion of new unit processes or data to those processes, there is no discussion of how to assess the spatial or temporal attributes of unit processes within the context of attributes used to define the system boundary or functional units. A challenge posed by bioenergy inventories is that in dynamic systems, the input and output flows from unit processes may not scale linearly, as additional processes, activities, and feedback loops create non-linear relationships (Collet et al. 2014). For example, soils can be both a sink and source of emissions depending on the combination of environmental factors and land management approaches influencing soil dynamics, including numerous non-linear feedback relationships (Núñez et al. 2012; Pawelzik et al. 2013; Petersen et al. 2013)

2.4 Life Cycle Impact Assessment (LCIA) – Assessing Commercial LCA Models

A number of commercial LCA packages are available that combine inventory databases with impact assessment software and calculate impact assessments based on midpoint and endpoints. These packages usually have user-friendly, single interface system. In North America, there are two widely-used, dedicated LCA models that focus on evaluating environmental impacts of bioenergy and particularly biofuels for transportation: GREET (Greenhouse gases, Regulated Emissions, and Energy use in Transportation) and GHGenius. GREET covers the full life cycle related to biofuels and bioenergy ('well to wheels'); it was developed in the U.S. by Argonne National Laboratory (Argonne National Laboratory, 2015). GHGenius is a greenhouse gas model developed for Natural Resources Canada based on an earlier version of the Lifecycle Emissions Model (LEM) (Delucchi, 2003). Both of these models are dedicated to vehicle and fuel combination scenarios. While the model boundaries are similar, GHGenius also considers the energy required to manufacture capital equipment associated with equipment and transportation (S&T2, 2005; 2009), generally ignored in other models because of the assumption that the production of farm equipment has a marginal impact relative to the overall life cycle.

There are also options in GHGenius to select a general geographic region to account for variations in fuel costs and production and include Canada, U.S., Mexico, and India, with further regional specificity available within Canada, where model conditions can be tailored for broad eastern, central, and western regions of the country. One LCA package being employed by researchers at the Government of Canada is SimaPro, developed by Pré consultants (2005) and designed to be a generic tool for evaluation of a range of processes, including transportation fuels. In the context of bioenergy there are several inventory databases included with SimaPro that can evaluate agricultural products and processes, including the Agri-Footprint and EcoInvent databases. Other proprietary or commercial LCA packages are structured in a similar way to SimaPro, including as TRACI, OpenLCA, GaBi, and Umberto. A significant difference between the software is the inventory databases included or available for each software package however a number of them share similar databases. For example, the EcoInvent database is accessible for SimaPro, GaBi, Umberto, and TRACI, and includes more than 10,000 datasets. The Agri-footprint database is available for both SimaPro and OpenLCA and contain inventory data for approximately 3500 products and processes specific to agriculture can be considered.

2.4.1 Current Model Limitations & Challenges

A number of articles have discussed the varied results that can occur in LCA depending on which software and what kind of system is being evaluated, highlighting the individual considerations practitioners should consider when choosing which software or method to employ. This is particularly important when different impact assessment methods can be applied to similar product system (Speck et al. 2015; Miller & Theis 2006). Each package includes a number of accessible databases, and can calculate impacts based on a number of different 'classes', generally as end-points or mid-points. Midpoint impact categories tend to relate to broad environmental themes such as climate change, eutrophication, or human toxicity (e.g. problem-oriented approaches), while endpoint impact categories tend to relate to areas of concern such as human health or natural resources impacts. Interpreting what these impacts mean at the farm-scale can be difficult though as these impact categories are aggregating inventory data from a variety of different scales. Which activities at smaller spatial and temporal scales contributed the most to those impacts are difficult to understand and so results can become difficult to compare across different LCAs that use different software packages.

Each of these LCA models and packages are limited when determining more site-specific inventory details. In the case of both GREET and GHGenius, there is a lack of detailed pathways to assess the impacts of biofuel feedstocks at the farm or field scale. Additionally, because of their focus on GHG emissions, other important and emerging areas of impact concern such as water use, water quality, soil quality, and land use changes are also ignored. In the case of the commercial inventory databases included with the LCA packages discussed, there are improvements in the detail of farm processes compared with GREET and GHGenius, however there is a limit to both the scale of that data and the number of datasets included. For example, in the Agri-Footprint database there is only one inventory value available for corn feedstock, reflecting five different countries; inventory units are all expressed as per kg units. Further, lignocellulosic feedstocks are not available in Agri-Footprint and are underrepresented in most of the databases used by the dominant LCA packages. While the OpenLCA database includes a value for willow feedstock production, they are based on European or global average values. Agricultural production values are only available for commercial food and feed crops in Canada, but even those are only based on Quebec data. The US LCI database has similar limitations in that there are limited lignocellulosic inventory available, often with limited geographical coverage. For example, while industrial corn production in the U.S. may be represented by a few dominant varieties, recent breeding programs focusing on SRC shrub willow over the last 20 years has resulted in more than 200 families (Serapiglia et al. 2008). Consequently, the volume and detail of life cycle inventory (LCI) data associated with agroecosystems and in particular emerging agroforestry products compared with traditional industrial LCI databases are not yet as extensive (Cherubini et al. 2009; Davis et al. 2008). The result is that not only are geography and crop type not included, but there is also no way to gauge how changes in on-farm or site-specific technologies, or land management approaches that change may affect the average values provided in those databases.

While the generic and broad packages can provide a level of understanding at a large scale, they are severely limited in their ability to guide land-management or farm-scale decision-making that may aim to improve feedstock production sustainability and reduce the overall impact signature of feedstock production. A recurring issue is that in many cases, data required to facilitate in-depth analyses at the farm or field scale are missing.

Fundamentally, there is difficulty assessing changes within the system at these fine spatial scales, and subsequently have little understanding about how changes in behaviour or technologies associated with land management are going to have an influence. There are also important ecosystem functions and services that agroecosystems can provide *or* disrupt within a landscape, which require analyses over specific temporal periods (Power 2010) that may not match the reporting period used. Gaps in understanding ecosystem impacts at fine scales may lead to misevaluation of processes that occur at the larger scales, reflected in LCA midpoints such as global warming potential (GWP), eutrophication potential (EP), acidification potential (AP), and fossil fuel use (FFU). Any desire to reduce the uncertainty of bioenergy systems are thus dependent on improving our ability to answer site-specific questions related to agroecosystem dynamics that previous approaches have been unable to answer or avoided.

2.5 A Need for Inclusion of Scale

Of particular relevance to geography when evaluating appropriate spatial and temporal data use, but notably absent from LCA standards reviewed in this chapter, is the concept of *scale*. The concept of scale helps to explain how the extent to which a problem is considered, and the resolution at which it is observed, may influence outcomes (Allen & Hoekstra, 1992; Levin, 1992). The lack of direction in managing spatial and temporal data resolution in 'data quality requirements' identified previously can enable practitioners to justify choosing data that only satisfies geographical extent; this limits the ability of practitioners to address geographical variations in the way that crops interact with local environments and different land management approaches. Manson (2008) discusses in detail the complications that can arise if one assumes that one scale perspective is more appropriate to observe a system at than all others, particularly when dealing with human-environment systems. This discussion identifies the extremes to which scale is affected by epistemological context, where a scale continuum exists somewhere between realism at one end and constructionism at the other. Landscape ecology faces similar challenges, where many aspects of ecological processes can change with the scale at which they are observed where larger scales are critical to context and smaller scales important in defining mechanisms of patterns observed (Allen and Hoekstra, 1992).

There may be links between different scales that have strong influences on overall results; the practitioner can overlook these and the data that is being collected and used in a single LCA may reflect very different spatial and temporal scales. For example, when understanding soil responses to different cultivation regimes, soil can be both a sink and source of emissions depending on the combination of environmental factors and land management approaches influencing soil dynamics, including numerous non-linear feedback relationships (Núñez et al. 2012; Pawelzik et al. 2013; Petersen et al. 2013). There is also increasing awareness about the important links between land management decisions, crop dynamics, and soil processes at sub-field spatial scales. Soil heterogeneity within a singular field has been shown to influence whether soil is a net sink or source of carbon at the same yield and residue harvest regimes (van Groenigen et al. 2011); this suggests that applying average residue removal rates may contribute to significant carbon loss within fields if these small-scale variations are not considered. In such situations, understanding what human-nature dynamics are occurring in each specific system requires the ability to view the system at a number of spatial and temporal scales.

Clearly, there is strong rationale to select data that reflects temporal and spatial specificity as the timing of inputs and outputs from agroecosystems can have different effects within the same space, or at the same time across different spaces. Agroecosystems - and most human-environment systems for that matter - rarely see processes occurring on the same spatial or temporal scales. To understand these systems, several different types of criteria may need to be assessed simultaneously such as long-term climate trends, mid-term harvest yields, or short-term changes to soil moisture. The challenge moving forward in modifying LCA approach therefore lies in finding common approaches to documenting and integrating the geographical attributes of processes and activities that occur at different spatial and temporal scales and have been aggregated a number of ways.

LCA in agroecosystems is complicated. Soil processes can happen at a very fine spatial scale over a very short period (m^2 and days), while biomass growth may be better understood at larger area and time scales (ha and months), and while land management itself may be best interpreted at the farm scale and take place over a year. Ultimately, land management decision-making is influenced by characteristics that are present over a range of spatial and temporal scales that may be much smaller than those used to view the majority of product systems in LCA and which might be important to look at (see Figure 2.3). Because LCAs can be performed at local, regional, or global scopes, it is imperative to engage in discussions of scale in order to reduce the impacts that competing interests at each level of analysis may create if other scopes are ignored.

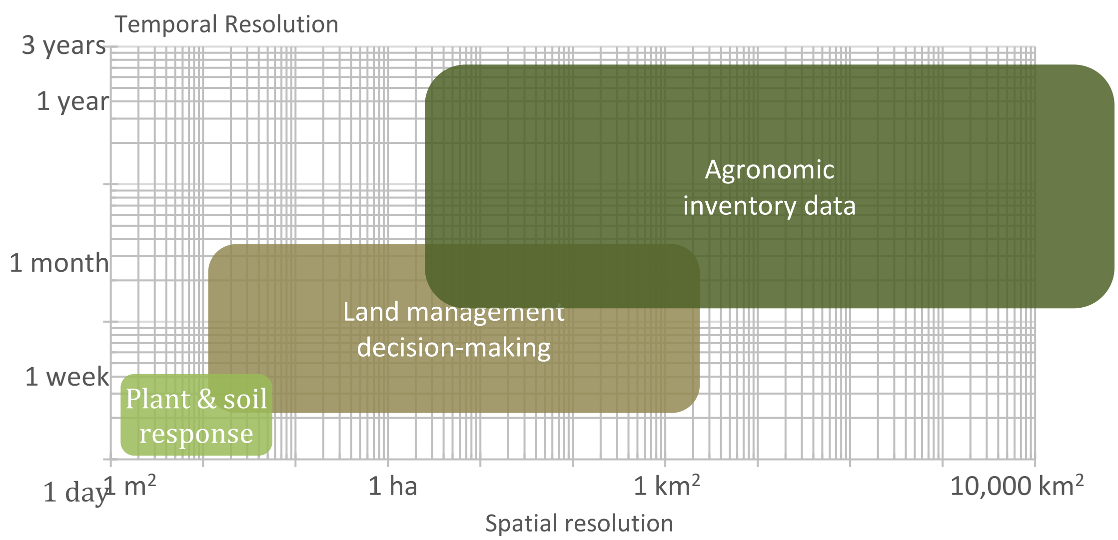

Figure 2.3. Ranges in spatial and temporal scales of agroecosystem data relative to system dynamics associated with biotic and abiotic responses and land management decision-making.

Temporally aggregated data can also limit the ability to understand seasonal variations of some impacts such as aquatic eutrophication (Conroy et al. 2010). The timing and rate of release of many emissions can have different effects on certain types of environmental impacts (Stasinopoulos et al. 2011). The tendencies of LCA to amalgamate inventory impacts into large singular compartments or production stages makes understanding variations in emissions from one time period to the next difficult, as there is no consensus on how to manage multiple emissions over varying time periods. Data inputs are increasingly heterogeneous across different geographies as climate, land-use, and local needs change; the use of singular averaged values ignore these small-scale nuances and limit our capacity to effectively evaluate future bioenergy scenarios.

LCA approaches which use aggregated data are unable to discern the combination of weather, land management, or local environmental variability may have contributed to the aggregated values being applied. Additional knowledge about these variables might be useful to consider when making decisions aimed at improving the environmental performance of feedstock production. At the field level, biomass production can be influenced by external climate-scale factors such as variations in radiation and rainfall that directly affect biomass growth and yield potentials (Horne & Grant 2009). An open product system means there are limits to land managers' ability to apply counter-measures to external environmental pressures. Irrigation may not be an option for some crops in arid or increasingly water-limited regions (Pfister et al. 2011; Strzepek et al. 2010; Evans & Cohen 2009) and the impacts of fertilizer runoff to external waterways may force crop cultivators to re-examine application rates (Scavia et al. 2014). The result is that not all external influences are going to have the same affect from one location to the next, requiring individualized approaches to system boundaries. Continued application of LCAs evaluated at large spatial and temporal scales clearly limits its ability to inform smaller scale feedstock decision-making or understand local influences on environmental impacts.

2.6 Implications for Resolving Land Use Change (LUC) Issues

Continued application of LCAs evaluated at large spatial and temporal scales clearly limits its ability to inform smaller scale feedstock decision-making or understand local influences on environmental impacts; these limits can have significant affects on the ability of LCA to inform increasingly important questions about land-use impacts of emerging bioenergy technologies. Given that land-use change (LUC) is the second largest contributor of global CO_2 emissions behind the use of fossil fuels (IPCC 2007) it is problematic, methodologically, that direct LUC is not addressed in LCA standards and indirect LUC (iLUC) is mentioned only once (discussed below). As examined previously, human-nature interactions on land used for biomass production, regardless of its end-use, creates direct interactions with soil, biota, and other environmental factors that has the potential to change those factors from past realities.

Both natural and human systems can act as drivers of change in agroecosystems; however, under circumstances where anthropogenic drivers dominate that change, understanding how constructed structures influence ecosystem functioning is critical to determining direct LUC. Direct LUC is not in and of itself an impact category, but rather an amalgamation of several impacts; these may include soil carbon and nitrogen changes, salinity changes, evapotranspiration and soil moisture changes, and biodiversity. These are all areas where changes to the land can result from agricultural practices and thus contribute to direct land-use changes. Thus, many of the impacts and dynamics that make up LUC as a collective may require small-scale spatial or temporal resolution data to effectively answer some land-use change questions. As a result, improving our understanding of direct LUCs requires an improvement in our ability to consider site-specific human-nature interactions and dynamics that is currently a challenge for current LCA frameworks.

2.6.1 Addressing Indirect Land Use Change (iLUC)

Much of the concern surrounding iLUC lies in the perception that when croplands are diverted to biofuels, the continued demand for those displaced crops are likely to increase commodity prices and incentivize converting existing land use elsewhere (e.g. forests, grasslands, wetlands) into agricultural land. Many of the early iLUC concerns raised focused on increased deforestation in South America as a result of increased corn-for-ethanol production elsewhere (e.g. the U.S.), advocating the need to include iLUC as part of the LCA impact assessment (Searchinger 2008; Fargione et al. 2008).

Assessing iLUC as a single impact category is problematic: there are challenges in the way iLUC is measured across different land uses, and causal relationships between land-uses become more difficult as the space and time between land-uses increases. There are also questions about how iLUC would be governed, even if the preceding challenges were resolved (Gawel & Ludwig, 2000).

Definitions in LCA standards inform practitioners that iLUCs, "…are those changes caused by market forces." (ISO 14040, 2006) but offer no real direction on how to measure or apply iLUC and so depending on which parameters are used to assess iLUC, very different results can manifest. For example, correlations between corn ethanol production increases and food commodity prices in the early and mid 2000's have been made (Pimental et al. 2009; Wallington et al. 2012) and a number of articles have proposed that all new crop production should have an iLUC indicator applied to their assessment as a result of these observations. Others have suggested that from 2000 – 2010 there was no significant decrease in agricultural exports from the U.S., suggesting that any correlations drawn between ethanol production and commodity prices were not entirely causal (Wallington et al. 2012). Yang & Suh (2015) also brought attention to the fact that between 2005 – 2009, the area of corn harvested in the U.S. increased by 1.8 million ha, but the area of cotton harvested decreased by 2.5 million ha. Ajanovic (2011) observed similar trends where although the share of bio-based fuels has increased, feedstock production and yields per area of land cultivated had increased proportionally. Each of these examples shows how assuming that new crop production may not always result in iLUC at smaller national or regional scales and that using absolute changes in biofuel or feedstock production as a means to measure links between land-uses is also questionable. These examples also highlight the problem in measuring and applying iLUC the same way across all geographical scales.

When a geographical perspective is used to explore iLUC challenges, there is reason to believe though that large-scale correlations drawn between bioenergy, commodity prices, and indirect land use change may not tell the whole story about what types of dynamics are actually happening at smaller scales. For example, DeFries et al. (2010) have correlated deforestation with commodity prices in Brazil, and Hausman (2011) showed that soybean prices are linked to field expansion. When sequential land-use was compared over time by Zilberman et al. 2010), they found that most of the deforestation occurring in Brazil was initially a direct result of rangeland conversion, while soybean cultivation tended to follow rangeland use once soil resources had been depleted. Macedo et al. (2012) observed similar land use patterns in the Mato Grosso region of the Amazon where 74% of soy production increases from 2001-2005 was into previously cleared pasture and increased to 91% of production from 2006 – 2010, and correlated with a significant drop in the proportion of cropland expansions (10% to 2%) that caused deforestation. In these instances, when land use trends are analyzed at finer spatial and temporal resolutions, they offer a different perspective than the observations made using much larger aggregated data.

There are also a number of articles that have conflicting reports about the causality of iLUC claims at the global scale as biofuels are not the only factor that can affect commodity pricing. Zilberman et al. (2010) make that point that the Conservation Reserve Program in the U.S., which incentivizes field transitions to forest land, might also incentivize deforestation elsewhere. While Hochman et al. (2010) identified the commodity spike from 2003 – 2008 as being influenced by biofuels, they concluded that demand for commodities was the major driver. Others found that increasing energy costs, devaluation of the U.S. dollar, and crude oil volatility were found to be the biggest drivers of commodity prices and as a result the additional costs of production also meant that those price increases were rarely being passed along to farmers (Armah et al. 2009). Indeed, recent work has identified that while the price of commodities and crop production are influenced by price at market, farmers will tend to switch to crops with less volatile prices that limits the uncertainty of their costs and minimizes financial risk (Haile et al. 2016). Similar phenomenon may be observed with indirect market forces associated with areas where local economies are highly coupled with agriculture or forestry compared with others dependent on global markets (Dong et al. 2013; Aistrup et al. 2013; Murphy & Lawhon 2011). What this and others say though is that the assumption that global market commodity prices are driving decisions about which crops to deploy are not necessarily true; again, there is a disconnect between the data we are using at large-scales and the decisions we are making at the farm or land-management scale.

Of critical importance, but absent from most iLUC discussions, is that indirect effects on other land uses can manifest in ways other than large-scale market forces. It has been shown that eutrophication of local waterways are an increasingly large concern in agriculture and certainly exhibit strong connectivity with neighboring ecosystems with closer proximity (Scavia et al. 2014). In Chile, shellfish from coastal waters that were adjacent to watersheds with tree plantations were found to be of reduced market value due to poorer biological conditions, a problem that has resulted in calls for management of regional land-use changes using more integrated socio-ecological approaches (Van Holt et al. 2012). Understanding the connection between these types of indirect effects though require better understanding of direct land use changes in the context of different spatial and temporal scales. For example, while nitrates and groundwater quality may be evaluated at catchment scales, the impacts from individual farms are likely to be lost in that data (Shabalala et al. 2013). Similarly, Buck et al. (2004) found that local land-use was important for smaller waterways while upstream land was likely to have greater influence on downstream larger waterways. Temporal variations in direct land use changes are also critical for understanding effects on land elsewhere as Shabalala et al. (2013) show that water quality can experience temporal changes as a result of both seasonal changes and activities from nearby farms. What this discussion reinforces though is that indirect land use change needs to be evaluated at scales smaller and more variable than global market scales, and that depending on the types of indirect effects we wish to evaluate, the spatial or temporal scale of observations may need to be altered to optimize accurate assessments.

What is also important to acknowledge from this review is that there are currently no universally accepted approaches to evaluating LUC, direct or indirect. It is also clear from the literature that assigning a singular LUC indicator or metric is likely to be inappropriate as LUC effects are not singular impact entities but rather an amalgamation of several different impacts that can manifest at a variety of spatial and temporal scales. Small-scale field management and local markets may not reflect trends observed at larger scales where spatial and temporal data are aggregated, and so our ability to inform land-management and policy decisions about indirect effects at more local scales may be limited. As a result, there are deeper social, economic, and environmental pressures that exist beyond large meta-data correlations as a means of large-scale decision-making. There is a need to be able to generate better direct land-use data at improved spatial and temporal resolutions so that direct and iLUC can be defined based on more geographically explicit terms. In such instances, small-scale data that retains its attributes in a large spatial extent but improved spatial resolution has the capacity to make more informed decisions regarding land-use at small-scales, and help determine the appropriateness of the conclusions we can make from the results we get. As identified, improving the site-specificity of LCA data is more than simply identifying different geographical locations and taking a measurement; it involves understanding that natural and human interventions can influence system dynamics and accepting that each may have unique spatial and temporal scales at which changes should be observed. In order to resolve issues related to geographical uncertainty of LCA data, better frameworks and ways of analyzing LCA are needed that can integrate spatial and temporal process attributes at different scales.

2.7 Key Findings and Recommendations

There are four key findings and associated recommendations that could help to improve the disconnect between geographical data and LCA components:

i. LCA standards do not provide direction in setting geographical parameters – this is left to the practitioner to interpret

There is need for guidance on how system boundaries are set from a spatial and temporal attribute basis. System boundaries of bioenergy product systems tend to include processes that exist over large geographical areas, however the detail at which those processes should be defined can vary depending on the objectives of the LCA. The 'Data Quality Requirements' section only advises practitioners to meet the geographical extent of product systems, and may misrepresent critical information about finer-scale processes. System boundary scales need to be more explicitly defined in the context of the objectives of the LCA. This way, assessments that wish to explore generalized footprints or screening studies can feel confident using large aggregated data, while those focused on site-specific issues such as changes in soil carbon will see that they may need to look for sites that exhibit similar characteristics to support the LCI development.

ii. Cut-off criteria may lead LCAs to overlook important impacts happening at fine spatial or temporal scales

Cut-off criteria need to include a discussion of the resolution that each system process should be assessed at, so that the spatial and temporal boundaries chosen to analyze data is transparent and cut-off decisions can be made based on data limitations, bias, or other criteria. Definitions for cut-off criteria are based on satisfying the geographical extent of the system or process but there are no discussions about satisfying the resolution of process dynamics. As a result, small-scale processes can be easily and justifiably ignored (e.g. soil carbon processes) by only addressing geographical extent. Including discussion of spatial and temporal scales in cut-off criteria would not inhibit these types of broad-reaching LCAs from being performed, but would require explicit descriptions about why processes where included or ignored in a more transparent way.

iii. Data quality requirement checks data do not explicitly address the geographical resolution of unit processes relative to system boundary definitions

There is a need for the goal and scope section to address appropriate geographical resolution of data and system flows in addition to their geographical extent. Data quality requirements are routinely called on throughout the LCA standards to provide direction on how to determine the appropriateness of inventory data, unit process inclusion or exclusion, and verifying that inventory and impact assessment criteria are satisfying the goal and scope of the LCA. If initial data requirements are never required to address spatial or temporal resolution of unit processes and material flows relative to the geographical extent of the product system, there is little opportunity for the data quality requirements to address whether downstream applications of the LCA methodology are being conducted appropriately. Including discussions of scale within the goal and scope would provide practitioners with a framework with which geographical mismatches between product systems and system processes could be addressed.

iv. LCA impacts are often aggregated from system processes and reported as midpoints and endpoints that can conflate geographic scales

There is a need to understand some processes and their associated impacts at different scales than used by many LCA studies and software. Interpreting endpoints and midpoints are difficult for complex systems, and so it may be more appropriate in some instances to set midpoints for specific subsystems that might better reflect appropriate temporal and spatial scales of analysis. Generalized impact categories, and in particular those that characterize large-scale impacts such as global warming impacts, may not effectively inform LCA practitioners about the small-scale impacts that may be important at field or landscape scales. It is clear that in many cases, data at finer temporal and spatial scales area required to answer outstanding questions related to land use changes that cannot be answered by using generalized impact categories. For example, eutrophication may be directionally and temporally dependent based on spatial locations of waterways and seasonal variation in nitrogen dynamics, while biodiversity impacts may be more important in some landscapes that have more critical habitat patch sizes relative to others.

2.8 Conclusions

The concept of scale is not currently a widely discussed topic within the LCA community. This review suggests that it needs to be addressed if geographical attributes are to be managed across different scales, particularly given the heterogeneous nature of biomass feedstock production across landscapes. Different input variables may reflect different spatial and/or temporal scales. There is currently a lack of direction on how to select appropriate spatial or temporal scales in LCA, and thus the impact of geography may be masked. Specifically, there are no guidelines to help select appropriate spatial and temporal resolution in the data. Over the longer term, LCA standards need to include spatial and temporal attribute information related to key components highlighted in this review, including product and system boundaries, life cycle inventories, and impact assessment reporting.

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Chapter 3 – Systematic Review of Life Cycle Assessments for Bioenergy Feedstock Production (2005 – 2015): Comparing Geographical Attributes of Systems, Parameters, & Inputs

Abstract

The challenge in consistently applying LCA is that the open boundaries and changing scales of biomass production differs considerably from the more controlled, place-agnostic industrial systems for which much of the early LCA approaches were designed. As the geographical scale and system dynamics of the bioenergy supply chain under study changes, so too do the land management decisions associated with biomass production in response to changing local conditions such as climate, soil, and native biodiversity. In an attempt to understand and ultimately resolve some of the discrepancies observed between study results, a systematic review of the variation in spatial and temporal scales associated with choices made in bioenergy LCA studies is conducted, using four feedstock crops of interest: corn, miscanthus, switchgrass, and willow. Patterns in choices made showed that the spatial and temporal scale at which the study was conducted often differed from the resolution of the data used, suggesting that common functional units may not qualify results between studies to be fully comparable. Several examples of where data has been scaled down and up from its source to the final impact assessment phase are illustrated. What is apparent about many of these scale transitions is that they are often not discussed or justified; this highlights gaps in LCA standards with respect to attributing data quality that need to be addressed in order to improve transparency and standardization, particularly when assessing environmental systems. Incorporating the spatial and temporal scales of attribute data may allow for improved comparisons between different LCAs, and support supply chain decisions such as crop selection, fertilizer applications, harvest times, and pre-processing technologies.

3.1 Introduction

Bioenergy systems and in particular biomass feedstock production can be highly variable across different geographies where climate, land use, and ecology can all influence system dynamics. A common method of assessing the impacts of bioenergy systems, life cycle assessment (LCA), therefore must address a number of dynamic issues across a range of political and economic interests. For many LCAs there continue to be gaps in inventories where data have yet to be generated for the variety of conditions and technologies that might be assessed in a given scenario and forces practitioners to either forgo analysis in their absence or use proxy data as an alternative (Canals et al. 2011). Agroecosystem characteristics are increasingly heterogeneous across different geographies as climate, land-use, and local needs change; the use of singular averaged values may ignore these small-scale nuances and limit our capacity to effectively evaluate future bioenergy scenarios. Differences in yields and production impacts are likely to vary depending on the location and approaches to feedstock production so using large-scale aggregated flows may lose some critical geographical information that might be important to keep track of at more local, geographically explicit levels of detail.

There have been numerous reviews highlighting differences between consequential and attributional approaches (Sanden & Karlstrom 2007; Thomassen et al. 2008), system expansion and allocation (Curran 2007; Ekvall & Finnveden 2001; Kauffman et al. 2011), or variations in inventory choices (Ekvall & Weidema 2004; Rebitzer et al. 2004; Frischknecht et al. 2004). To date, no study has systematically looked at how comparable LCA studies are from a geographical perspective. Of particular interest is how well the scale(s) of data input compares to the scale of the system being assessed. It is also important to know how issues of scale are managed through the LCA process; knowledge along these lines may improve (or hinder) comparisons with other

LCAs. In order to begin to develop better direction in handling varying spatial and temporal attributes, this study seeks to understand where these scalar challenges can manifest within existing bioenergy LCAs.

This study seeks to help advance discussions between dynamic LCAs and geography by performing a comprehensive literature review of bioenergy LCA literature over a ten-year period (2005-2015), with a particular focus on summarizing the spatial and temporal attributes of feedstock production data used at each stage of the assessment. The review first identifies some important geographical differences that might affect biomass production performance at the farm- or field-scale and summarizes some of the key spatial and temporal attributes from the four LCA stages (Goal and Scope, LCI, LCIA, and Interpretation). It is hypothesized that LCAs that seem to be comparable may have distinct differences in the spatial and temporal attributes considered. The lack of direction in how to document and account for those attributes contributes to ongoing challenges in attempts to compare and evaluate the results from those LCAs.

3.2 Review Methods

This review considers ten years (2005 – 2015) of published, peer-reviewed LCA work related to bioenergy and biofuels worldwide, developed by performing a comprehensive literature search through the Web of Knowledge. Two topic keywords were entered to find LCA-associated articles within the searchable timeframe: "life cycle", and "LCA". The selection of studies was refined to consider one of four biomass feedstock crops of interest to the established and emerging bioenergy and biofuel sector in Canada. Corn is one of the dominant bioenergy and food crops globally, and including this feedstock meant that the study could contrast temporal differences between land management applications for cereal crops to those of perennial grasses (miscanthus, switchgrass) and semi-annually harvested short rotation woody coppice (willow). The keywords applied to the literature search were "corn", "miscanthus", "switchgrass", or "willow".

There were 1,246 articles returned across all permutations of the four feedstocks. Articles that were reviews, letters, and conference proceedings were not included as the focus was on studies that were performing a LCA based on ISO standards or life cycle principles. Articles whose focus was on economic considerations rather than environmental impact categories were excluded. In addition, LCAs that did not include feedstock production within their system boundaries were excluded, as the focus of this review was to compare how feedstock production data differed at spatial and temporal scales across the studies reviewed. Ultimately, 76 unique studies were identified as suitable for this review. To summarize aspects of the goal and scope components, as described in the previous sections, six different metrics were compared and summarized; they are listed in Table 3.1. Five inventory components were also summarized; they are listed along with descriptions of criteria used to summarize them in Table 3.2.

It should be noted that some studies considered more than one feedstock, which is why the total number of studies reviewed (76) is lower than the number of times a particular feedstock is considered (96). Similar trends exist throughout each of the categories assessed where each study may have assessed or included more than one of each study characteristic summarized, so the number of studies (n) included in each evaluation is provided where appropriate. The following sections summarize the highlighted trends from those entries, but a detailed summary of all studies reviewed can be found in Appendix 3.1 of supplemental information in their entirety.

3.3 Results & Discussion

3.3.1 Goal & Scope Components

On a global scale, about 65% of the studies assessing transport fuels were carried out in North America, while approximately 84% of studies assessing heat or electrical energy originated out of Europe. This reflects North America's focus on greening transportation fuels and Europe's focus on reducing greenhouse gas emissions. Studies that focused on transport fuels also tended to be associated with corn as the primary biofuel feedstock; the majority of transport fuel related articles (54%) and North American articles (57%) considered corn. European studies, on the other hand, have tended to focus on perennial lignocellulosic feedstocks where 57% of miscanthus and 63% of willow crops were assessed as feedstock options to feed into heat and electricity end uses. The distributed tendencies of LCAs are illustrative examples of how preferences for feedstocks for particular product end uses can direct the frequency of feedstock use geographically.

Table 3.1. Summary of Goal & Scope components and the metrics used to summarize and compare them across studies is described.

Metrics	Descriptions
Product System Scope	Scopes listed as each study describes, such as 'Cradle-' 'to-Farm' or 'to-Grave', and 'Well-' 'to-Wheels' or 'to-Tank'. For studies that did not use generalized scope terms and instead described their supply chains, the term 'Described' was used to summarize the scope for comparisons
Feedstock(s)	Feedstock crop(s) used for energy product(s)
Functional Units	Functional units are stated as they are described explicitly in each review. Where units are not clearly identified, articles are summarized as 'not explicitly stated'.
Functions of Product System	For scopes that end at the Farm Gate, the type of product being transported from the farm is stated. For scopes that extend beyond the farm, the intended end-use is summarized (e.g. a '1 L ethanol' F.U. would be summarized as 'Transport Fuel'
Supply Chain Location	Three levels of geographic area are used: Continental, National, Regional/ Site-Specific area or location(s)
Impact Categories	Impacts are summarized as including 'GWP/GHG emissions', 'Energy Metrics' (e.g. yields, balances, ratios), or 'Other impacts' (e.g. eutrophication, acidification)

Table 3.2. Summary of Inventory & Impact components and the metrics used to summarize and compare them across studies is described.

Metrics	Descriptions
Spatial Attributes of Yield/Farm Data	The extent (total area) of farm yields as well as the resolution (smallest unit area) that yield and farm data are reported at are summarized
Temporal Attributes of Yield Data & LCIA	Temporal period of yield data collected and the temporal length of the LCIA.
Feedstock Yield Values	Feedstock yields assumed/measured where they are available.
Soil Carbon Assumptions	Where soil C is not mentioned, studies are summarized as 'not explicitly stated'. Studies that justify ignoring soil C based on study-specific assumptions are summarized as 'No change is assumed'. Studies that include soil C changes or emissions are listed as either 'linear emission/ sequestration factors' (e.g. IPCC emission factors), 'direct measurements' (e.g. field trials), or 'modeled changes' (e.g. software or models such as DAYCENT or AMG)
Land Use Change Approaches/ Assumptions	- Studies that identified considering direct or indirect LUC (or both) are listed as such. The approach used to represent this change is listed immediately after such as 'total area of land for cultivation', or 'soil carbon changes'

3.3.2 Functional Units

Of the 76 studies reviewed, 16 did not explicitly state the functional unit being used. Upon close review, it was sometimes possible to interpret the intended FUs in these works, but as the number of systems being analyzed increased, teasing out common reference flows became difficult. Of those articles that did explicitly state their FU, 14 of them compared more than one FU; the fact that more than one FU was possible exacerbated the process of teasing out reference units from those studies that did not explicitly state them. The ability to understand what reference units are being considered is compromised when they are not clearly stated, which is a continuing problem in LCA studies. Part of that issue could be whether practitioners are referencing ISO 14040:2006 or ISO 14044:2006 standards. Within the ISO 14040 standards, the functional unit is listed as a component of the scope but no language is used to state that it needs to be explicitly described, instead it simply reads, "The scope includes the following items". The ISO 14044 standards on the other hand preface the list of scope items by stating, "the following items shall be considered and clearly described". There is no apparent reason for this discrepancy and it seems that it would be prudent to amend ISO 14040 standards language to ensure that the scope items, and in particular the functional units, are explicitly stated or described.

Explicitly stating the functional unit however was not necessarily a guarantee of being easily compared with other LCA results, largely due to the variety of nuances used to describe them. While there were certainly FUs that were similar, of the 50 articles where a FU was stated, there were 62 uniquely worded phrases to describe those FUs (Appendix 2). Many of the unique FUs found in these studies were used to report on a more limited set of common products or services, such as boiler heat production (Godard et al. 2013), biofuel processing capacity (Eranki & Dale 2011) or supply of heat to residential households (Kimming et al. 2011). A significant number of studies had similar FUs but chose to tag the FU with information about the size of the supply chain being assessed, or about specific products associated with that system. For example, Yang & Chen (2013) used the FU '1 ha' but there were several variations on that FU in other studies, such as '1 ha of production' that focused on a cradle-to-farm gate scope (Murphy & Kendall, 2013) or '1 ha of willow' (Gonzalez-Garcia et al. 2012a), or 'production of chips from 1 ha' (Gonzalez-Garcia et al. 2012b). Other studies made the FU country-specific, for example using '1 ha Danish agricultural land' which effectively constrains the study within a unique spatial area (Tonini et al. 2012a, 2012b).

In response to the lack of clear direction on to how to address spatial and temporal attributes, many studies have opted to tag geographic information within the FUs themselves (Table 3). While some articles incorporated time with space such as '1 ha of land for one year' (Brandao et al. 2011; Styles & Jones 2007) or '1ha/100yr' (Lettens et al. 2003), others focused solely on space including '1 ha land' (Fazio & Monti, 2011; Kauffman et al. 2011) and '1 ha of crop production' (Goglio & Owende 2009). There were also some articles that combined spatial and temporal attributes of their functional units with a specific impact category or indicator such as Hoefnagels et al. (2010) that incorporated energy use by applying the functional unit 'MJ_{prim}/ha' or Schumacher et al. (2010) that used 'energy yield per hectare' and 'energy yield per year'. Although many of these articles compared similar feedstocks or farm-level technologies, the ability to compare results between studies that used FUs tagged with different geographical attributes required deeper analysis of underlying methods and assumptions, clearly limiting comparability between LCA studies.

3.3.3 Product System Scopes

Of the 15 articles that explicitly used a 'cradle-to-farm gate' scope, the majority of them (61%) used raw feedstock mass or volume as their FU emphasizing the focus of those scopes on farm processes and feedstock supply. These results appear to confirm an assumption often made regarding the use of 'cradle-to-farm gate' scopes which assumes that they provide information that can feed into larger supply chain scopes that can eventually be used to assess final end-products or bioenergy use that requires biomass feedstocks (Bessou et al. 2013). The actual spatial extent (area) providing feedstock supply is highly variable across the studies reviewed. For example, Styles and Jones (2007) consider national averages of miscanthus and willow cultivation in Ireland from a region >30,000 ha, while Gonzalez-Garcia et al. (2012a) use pooled harvest data from 9,048 ha of Swedish land. By contrast, Gonzalez-Garcia et al. (2013a) use farm-specific data (<100 ha) from two locations in the Lombardy region of Italy, and Goglio et al. (2012) focuses on field trials from a 0.36 ha area plot in Tuscany, Italy. The spatial area that supply chain scopes consider sometimes include the distance that feedstocks travel to arrive at biorefining or product end-use locations, varying from 20 to 100 km radiuses (Kim et al. 2009; Murphy et al. 2013). This highlights the limitations of describing system boundaries using terms that are spatially arbitrary, and is a likely reason that a significant proportion (34%) of articles reviewed chose not to use 'cradle-to' or 'well-to' terms. Common terms used to describe the scope of LCAs may only be describing the length of supply chains, rather than capturing any sort of spatial or temporal scale, and in fact may confuse whether some studies are actually of a larger spatial or temporal scale than other studies.

3.3.4 Impact Categories

Life cycle assessment generally summarizes environmental impacts in specific impact categories, often described as midpoints. In 15 of the 76 studies reviewed, GWP and GHGs were the only impact categories assessed. In another 16 studies, these impacts were combined with energy metrics such as energy yields and ratios. This means that nearly half of all studies reviewed (41%) were limited to global warming and energy impacts which mirrors trends observed in previous work (Bessou 2013; Curran 2007; Goglio et al. 2015). Attempts to quantify changes in soil carbon were not included in about 42% of the LCAs considered in this review; this corroborates the observations of other authors (Djomo et al. 2011; Bessou et al. 2013; Brandao 2011).

In those studies that did include soil carbon, about 16% assumed that soil carbon levels would remain unchanged. This assumption was occasionally justified because land use for the area under consideration was not anticipated to change and standard agricultural practices would continue to be applied (Buratti et al. 2013; Blengini et al. 2011). Other justifications for assuming that no net change in carbon would occur were related to management approaches associated with specific crops. This included conservative estimations about no-till perennial crops such as willow (Budsberg et al. 2012; Heller et al. 2003); similarly, corn stover removal rates from corn crops are assumed to occur at sustainable levels (Kaliyan et al. 2011; Spatari et al. 2005). Difficulties in accounting for the influence of cultivation activities and environmental factors on soil carbon and the resulting limitations on available data for changing crops or conditions were also identified as a reason for ignoring soil carbon dynamics (Gonzalez-Garcia et al. 2013a; 2012a; 2012b). This reinforces the constraints that limited inventory databases have for robust assessments of agroecological systems. Each of the approaches to managing soil carbon described above illustrate how cut-off criteria are used to justify the exclusion of soil carbon dynamics in bioenergy LCAs. An additional trend observed among the studies reviewed was an increasing number of articles over time which utilized LCA software to evaluate midpoint categories such as carcinogens, smog potential, respiratory effects, and ozone depletion (e.g. Ashworth et al. 2015; Bergman et al. 2015; Hong et al. 2015; Jeswani et al. 2015; Yang & Suh 2015). While analysis of these categories allow for comparable studies with fossil fuel alternatives, they provide little direction for biomass feedstock producers in terms of guiding improved performance, and may not always be relevant for regional assessment of direct impacts on local environments.

3.3.5 Inventory Components

3.3.5.1 Yield Inputs

The range of biomass yields found among the LCA studies reviewed, for those studies where an average annual yield value was given or could be calculated, varied the most in corn and the least in willow (Figure 1). There were more studies where corn yields were available (n=22) than switchgrass (n=13), willow (n=12), or miscanthus (n=12) and consequently the number of different locations that corn yields were obtained for likely contributed to the variation in yields observed in corn. The expectation entering the review was that geography, and in particular the latitude at which crops were grown, were likely to be the most important determinants of yields. Upon initial examination, this seemed to be borne out by data used in the studies, particularly for corn yields. The maximum corn yield values of 70 t/ha/yr (for two harvests in a single year) (Blengini et al. 2011) and 51.5 t/ha/yr (Buratti et al. 2013) were both observed in Italy. The minimum average corn yield of 7 t/ha/yr was observed in Canada (Pelletier et al. 2008).

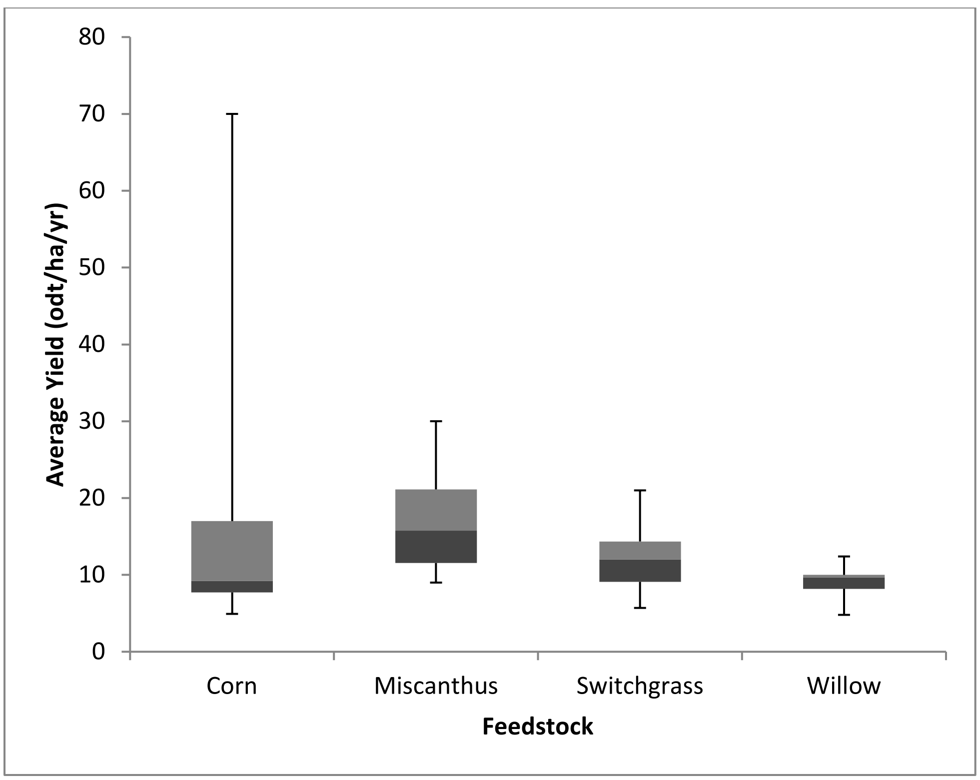

Figure 3.1. Box plot distribution of average yield values used for each of the four feedstocks, for those articles reviewed where yields were provided. The median is defined at the separation between light and dark portions of each box with the lower box representing the first quartile and the top of the upper box representing the third quartile position.

When yields were examined in more detail, the variations in yields could not be easily correlated to geography. For example, while Italy produced the largest yield, it also produced one of the lowest corn yields at 8.6 t/ha/yr (Goglio et al. 2012), comparable to yields in Canada of 8.3

t/ha/yr (Hussain et al. 2011). Additionally, corn yields from Germany were reported to range between 20 t/ha/yr (Schumacher et al. 2010) and 43 t/ha/yr (Dressler et al. 2012). The range in corn yields from Germany also highlight the variation in yields that can exist within the same region of origin. Similarly, corn yield values used in the U.S. Midwest ranged from 7-8 t/ha/yr (Kim et al. 2009; Geyer et al. 2013) to 13 t/ha/yr (Murphy & Kendall 2013). Other species were also highly variable; miscanthus yields from the U.K. ranged from an average of 9 t/ha (Hillier et al. 2009) to 14.4 t/ha (Brandao et al. 2011), while switchgrass yields in the U.S varied from 13.9 t/ha (Geyer et al. 2013) to 21 t/ha (Eranki & Dale 2011). Even willow, the feedstock which varied the least, was observed in Sweden at yields between 4.8 t/ha (Gonzalez-Garcia et al. 2013b) and 10 t/ha (Gonzalez-Garcia et al. 2012b). The higher correlation between willow yields is likely a function of the fact that 3 of the studies were from the same lead authors who compared the same general area and the same yield values, and that most literature and field trials for willow yields originates from northern Europe and New York State.

3.3.5.2 Geographical Attributes of Yield Data

Of particular interest when including temporal scale in a review of biomass yields is what effect, if any, the scale has on the yield values used by each study. When comparing two articles from Italy, Goglio et al. (2012) use a 17-year yield average (1985-2002) of 8.6 t/ha/yr, while Buratti et al. (2013) use a single year average (2010-2011) of 51.5 t/ha/yr. Because nearly every LCA article reviewed compiles production data into annualized values for impact assessment calculations, it is difficult to know what temporal periods are being represented. The disconnect between inventory values and space-time variability not only limits understanding yields; several other assumptions related to temporal aspects of production are ignored through aggregation, including patterns of crop establishment or tilling (discussed in Whitman et al. 2011; Kim et al. 2009).

No study in this review explicitly described the attributes of the yield data collected from sources; 22 studies did not include original spatial attributes, and 31 of them did not include original temporal attributes. Some of the studies could not summarize these attributes because the secondary source for the data was either inaccessible or in fact depended itself on another secondary source. When the spatial extents of yield source data were available and compared with the spatial extent that each LCA study represented there was often a difference. Of the 51 studies that provided a sense of the spatial extent from which yield data was obtained, only 11 used an area less than 300 ha is size; most were obtained from large scale areas and aggregated as average yields per hectare. For example, 12 studies aggregated yield data from national (e.g. Murphy & Kendall 2013) or trans-national averages (e.g. Brandao et al. 2011). Another five studies applied yield from provincial (e.g. Yang et al. 2012), state (Renouf et al. 2008), or large regional averages such as those for the U.S. Midwest and Corn Belt regions (e.g. Han et al. 2013; Kaliyan et al. 2011; Searcy & Flynn 2008).

What wasn't always clear from yield averages was the resolution or distribution of production within those larger spatial extents. Feedstock cultivation in highly productive or agriculture-focused areas, such as the U.S. Midwest, are likely to be more densely distributed per area of land. In this case, total biomass supplied for counties in these regions (explored by Kim et al. 2009) are likely to have different supply chain dynamics compared to less-productive areas. Missing resolution data also made understanding the relative use cultivatable land difficult, as highlighted by Geyer (2013).

Only one study scaled inventory data up from a plot smaller than 1 ha (Goglio et al. 2012). Two articles did not scale inventory data but totalled all the biomass supplied from a defined area (Buratti et al. 2013). A few other studies scaled yields without using a spatial component in the inventory. These studies recorded the total amount of biomass supplied to a refining plant per unit of time (e.g. Searcy & Flynn 2008), or the total amount of feedstock required to produce a particular unit of product produced (such as the mass of feedstock per mass of ethanol, e.g. Han et al. 2013). Many studies in the latter category focused on supply chain variables separately from the feedstock production stage, and so total product and raw materials available for supply were more important to focus on than feedstock production efficiencies per area of cultivated land.

3.3.5.3 Soil Carbon Inventories

Slightly less than half of the studies reviewed (42%) included some sort of soil C process approach. Of those that did include soil carbon, about half used a linear emission or sequestration factor (n=16) and half used a modeling or soil carbon accounting software approach (n=16) (Figure 3.2). The majority of the studies using linear approaches relied upon IPCC emission factors to assess the contribution of generalized land-use types towards global warming potential (GWP). By comparison, a few studies used linear sequestration factors to represent the ability of perennial crops to contribute to increasing soil carbon (Brandao et al. 2010; Cherubini & Jungmeier 2010; Godard et al. 2013; Styles and Jones 2007). Perhaps recognizing the importance of soil carbon to the overall system, 16 articles approached soil carbon process by using some sort of modeling application, for example DAYCENT (Adler 2007; Kim et al. 2009) or AMG (Godard et al. 2013). Even in bioenergy LCAs that use dynamic soil assessment methods though, large-scale global, national, and regional system sizes tend to dominate, a trend also observed by Goglio et al. (2015).

Discussion of aboveground residues was exclusively limited to studies that evaluated corn as a feedstock; in 15 of those studies residues and corn stover were never discussed. Another five articles explicitly stated that residues were not included as part of the harvest, instead leaving them on the soil for fertilizer purposes. There were seven articles that included stover removal as part of biomass yields, but did not indicate the proportion of the total residues on the ground that remained. This made it difficult to interpret whether stover was being left on the ground for future soil carbon inputs. The remaining 13 articles included stover as part of the harvest and ranged from 15% (Whitman et al. 2011) to 90% (Buratti et al. 2013) removal rates. No real pattern could be discerned, among studies that included stover removal rates, in terms of how soil carbon changes were approached. Four of those articles did not discuss soil carbon, three assumed no change, two used linear emission factors, and only four used some form of more dynamic model or software to evaluate soil carbon changes (as shown in Figure 3.2).

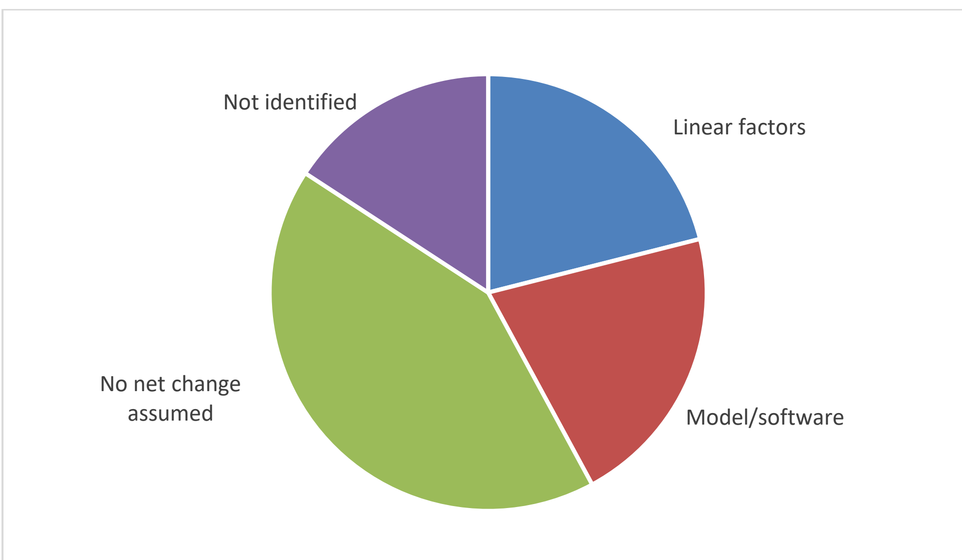

Figure 3.2. Distribution of methods used to assess soil carbon among all studies reviewed (n=76).

How soil carbon is treated in feedstock production analysis is of particular importance to the environmental performance of feedstock cultivation, yet is missing from a significant number of LCAs observed both in this review and in others. For example, in a recent review of LCAs focusing on perennial crops (Bessou et al. 2013), which included perennial feedstocks such as Jatropha and sugarcane from more tropical geographies, observed some of the same general trends. There were significant gaps in soil carbon accounting approaches, suggesting that if feedstocks compared was to be expanded, similar trends are likely to be observed. In Wiloso et al. (2012), the approach to residue treatments was inconsistent in looking at both soil carbon and yields, making it difficult to see how residues are connected with either soil carbon or feedstock yields.

Top-down inventory approaches that generalize soil carbon processes and impacts create disconnects between site-specific residue management, crop yields, and their influence on soil carbon dynamics. The lack of farm-level detail resolved during impact assessments was found to have a trickle-down influence on the ability of LCAs to understand more complex system concerns such as soil carbon accounting and land-use change where each of those components required different scales of observation to resolve better understandings of each relative to the level of detail that most articles were assembling field-scale inventories. One of the conservative approaches often applied in LCA is to assume there is no change in soil carbon due to the assumption that changing improved land management or crop types are likely to actually improve soil health. Several studies have observed no change in soil carbon storage under reduced till systems (Angers et al. 1997; Powlson et al. 2012). Dupont et al. (2010) showed that even under reduced till systems, conversion of grasslands to annual cropping will reduce soil carbon after a three year period, further emphasizing the importance of understanding land-use change through time. Blanco-Canqui & Lal (2007) showed that while no-till systems may increase carbon concentrations in some surface soils, the whole soil profile remains largely unchanged. These results identify the disconnect between residue management, crop yields, and soil carbon dynamics and illustrate the problem with trusting generalized emission factors or sequestration factors when it comes to estimating soil carbon impacts across a range of geographies.

3.3.5.4 Land Use Changes

Land use change (LUC) is another important factor to consider within LCA, but only 24 of the 76 LCAs considered in this review addressed direct or indirect land-use changes. Previous land-use decisions were a significant factor in determining whether LUC was included (Table 3.3). For example, the assumption that no LUC would occur was often applied when feedstock production was assumed to be sourced from land on which similar or identical crops had been grown historically (sometimes for as long as 60 years, which assumed that change was unlikely) (Buratti et al. 2013; Kim et al. 2009; Pelletier et al. 2008). Whether the changes assumed to be unimportant were soil changes, market changes, or some other land use indicator though was not always identified.

Table 3.3. List of the range in approaches used to address direct and indirect LUC among all LCA studies reviewed and the frequency they were applied among articles.

Land Use Change Approach	Number of Articles
Direct LUC - Area of Land Being Used	2
Direct LUC - Carbon emission factor	1
Direct LUC - Land Use compared to alternate crops/scenario	3
Direct LUC - Modeled Carbon Changes	7
iLUC - Carbon emission factor	4
No LUC - Similar or improved land/crop use assumed	7
Not Identified/ Clear	52

In many cases, there has been a tendency to consider potential changes to soil carbon and use that as a proxy for whether LUC is something to be considered. Examples of this include Budsberg et al. (2012), who assume that the introduction of perennial crops would lead to an improvement in soil carbon and thus chose to ignore LUC. Additionally, 100% of the studies in this review using soil C emission factors also used emission factors to address LUC; similarly, 100% of studies that modeled soil C change as a way to address LUC used the same approach for soil C processes. Ultimately, this means that there is a tendency within LCAs to treat soil carbon processes and LUC (or indirect land use change) as the same thing. Part of the reasoning behind this approach is likely due to observations that conversion of land into agricultural use has contributed to a 20-25% loss of soil carbon globally (Davidson and Ackerman 1993). While changes in soil carbon certainly contribute to land use changes, there are a number of other significant impacts associated with LUC that are potentially ignored if the maintenance of soil carbon levels are assumed to mean LUC is not relevant.

3.3.6 Impact Assessment Results

3.3.6.1 Temporal Scale Trends

When the length of time that each LCA assessed potential impacts was compared across each of the studies considered in the review. There was a clear trend towards using shorter time periods; 52 of the 76 (69%) articles calculated impacts for a one year period. Often this was because of the need to compare results to a single year value from annual crops such as corn, or to annualize yields and impacts for comparison with a reference system such as traditional fossil fuel alternatives. There were 14 studies that assessed impacts for temporal periods of 7-20 years, and 9 studies that considered longer periods (21 – 100 years). All of the LCAs that evaluated periods longer than one year did not account for variability in impacts from one year to the next.

While there were some articles that listed variations in production inventories between years, particularly those assessing perennial crops, those inventories were totalled and divided by the total number of years being assessed rather than assessing impact variability between years (e.g. Murphy et al. 2013). Other articles though applied impact assessments to multiple years by extending the time horizons of GWPs, however many of these were still assuming consistent impacts from one year to the next that affected climate change indicators. In general, while some articles assessed multiple year periods of impacts, they were often being scaled up from inventory values that were listed as an annual average, devoid of the inter-annual variability that might have gone into that average.

Several studies modeled environmental impacts over larger areas than the inventory values represented; inventory values were often represented at resolutions of one hectare. This has been observed in previous work, where most LCAs used average values at small local scales and scaled up to larger scales in a single step (Gerber et al. 2011; Raugei & Ulgiati 2009). The opposite is true when scaling trends in data are transitioned from yield sources to the inventory. This suggests that there are multiple points within the LCA methodology where scales in data can be manipulated; data scales should be identified in studies clearly.

3.3.7 LCA Comparability

To illustrate trends in impact results, and to understand the appropriateness of comparing results between studies with similar product systems and functional units, LCAs that assessed corn in the U.S. were considered. There are 25 studies available on this topic; this group of articles provided a larger sample size and more appropriate temporal scales to compare results, relative to studies that examined other feedstocks. Studies with common product system scopes and functional units were compiled. From the 25 studies, there were two different FUs, associated with two different LCA scopes, found to be the most common of those used. Each of these are discussed below.

The first group of articles (n=4) compared were those that performed an assessment from a cradle-to-farm gate perspective and evaluated the GHG emissions per mass of feedstock produced. On the surface, the articles appeared the most comparable as they shared common crop types (corn), country of origin (U.S.), temporal period of impact assessment (1 year), and common units to assess the GHG emissions per unit of feedstock produced at the farm gate (kg CO_2e/kg grain or stover). When the spatial and temporal attributes of input data are compared in addition to each article's treatment of soil carbon and land use changes, however, the underlying details are quite different. The spatial area for which source data originated included 280 ha (Eranki & Dale 2011), county averages (Kim et al. 2009), and an average for the U.S. corn belt region (Searcy & Flynn 2008; Xue et al. 2014). The temporal period for which those yields were compared was also different, ranging from one (Eranki & Dale 2011) to four year averages (Kim et al. 2009). The time period covered by Searcy & Flynn (2008) and Xue et al. (2014) was not clear from the individual studies. While Eranki & Dale (2011) did not consider soil carbon or land use changes, while others assessed soil carbon and ignored land use changes, by using a dynamic soil carbon model (Kim et al. 2009), or by assuming that soil carbon did not change (Searcy & Flynn (2008). Xue et al. (2014) did not explicitly discuss soil carbon changes. The range in characteristics of input data sources likely contributes to the range in impact results (-0.854 to 0.82 kg CO_2e/kg grain) which span both a net gain and net loss of carbon emissions. The results are not a critique of the approaches used in any one of these studies, but highlights the underlying geographical differences that are implied by the assumptions made by each study in working to achieve their desired goals. These differences should be taken into consideration when trying to compare the results between individual studies.

The second group of articles (n=5) considered were those that assessed systems from a cradle-to-grave perspective and evaluated GHGs per MJ of ethanol produced. Again, the articles appeared comparable, sharing all the same characteristics of the first group of articles except that impacts were evaluated based on the GHG emissions per unit energy of ethanol produced. The spatial and temporal attributes of data vary significantly from one LCA to the next. The spatial attributes of yield source data ranged from county (Adler et al. 2007) and state averages (Liska et al. 2009) to larger averages from the Midwest (Han et al. 2013; Kaliyan et al. 2011) and the continental U.S. (Wang et al. 2011). Similarly, the temporal period from which yields were obtained ranged from a single year (Han et al. 2013; Liska et al. 2009) to a decade (Kaliyan et al. 2011; Adler et al. 2007). While some articles did assess changes to soil carbon using dynamic modeling approaches (Wang et al. 2011; Adler et al. 2007), the remainder either ignored soil carbon changes (Han et al. 2013) or assumed that no change in soil carbon was likely (Kaliyan et al. 2011; Liska et al. 2009). Most of the articles did not discuss land use change in any capacity, but those that did chose distinctly different approaches that included measuring the area of land being used (Adler et al. 2007) and evaluating the changes to soil carbon as a direct LUC metric (Wang et al. 2011). The variance in the underlying attributes of each of these studies most certainly contribute to the variations in the performance of ethanol in terms of GHG emissions (from -15 to +76 g CO_2e/MJ ethanol). This makes comparisons of their results difficult despite the common regional areas and crops used among the product systems compared.

3.4 Limitations and Biases

The search criteria used in this review created a bias towards those feedstocks and definitely resulted in LCAs from regions where those crops were grown. In terms of the effect on overall results in this review, however, it is not believed that the overall results are likely to be influenced by this bias. The primary reason for this is that many of the articles reviewed (n=27) compared feedstocks in addition to the four summarized in this review, including wheat (e.g. Goglio et al. 2012; Fazio & Monti 2011), triticale (e.g. Buratti et al. 2013; Godard et al. 2013), cotton (Yang & Suh 2014), or rapeseed (e.g. Brandao et al. 2011; Hoefnagels et al. 2010). The approach to attribute scaling and data management were similar across all feedstocks.

3.5 Key Findings & Recommendations

The main findings of this review revealed that the spatial and temporal scales at which key components of bioenergy LCA studies are defined and evaluated are inconsistent between studies, even those that considered similar product systems or feedstocks. These results affirm the original hypothesis of this review, which was that while many bioenergy LCAs may compare similar feedstocks and processes, the underlying geographical attributes of data and the representative system being assessed are very different. This is one of the main reasons that comparisons among LCA studies are difficult and sometimes inappropriate. Although LCA standards emphasize the identification of the geographical extent of the product system and inputs, they provide no direction on how to define the geographical resolution of subsystem processes. They also provide little direction on how to manage spatial and temporal attributes of other important LCA components such as functional units, system boundaries, and inventory inputs. To improve the ability of LCA frameworks to evaluate dynamic environmental systems, five key findings and recommendations that can be derived from trends are provided:

i. Functional units are not explicitly described in all LCAs that consider bioenergy products or services

LCAs involving supply chains that include intermediate processes or products derived from or in environmental systems should be advised to explicitly state their functional unit. The existing ISO 14040 standards should also consider altering functional unit definitions to make it more clear that they should be explicitly defined, particularly given that LCA results of non-biomass energy products may be looked to for comparison with bioenergy LCAs in future policy or land-use decisions. It is possible that practitioners following ISO 14040 standards, which does not explicitly advise stating the functional unit like the ISO 14044 standards do, and may be responsible for some functional unit exclusions. The wide range of products and end uses possible for bioenergy and biomass feedstocks means that product systems and functional units may be comparing products or services that have undergone several transformation processes, far removed from previous environmental processes. The need to explicitly define their chosen functional unit may not appear as important as those studies that focus on more environmental-oriented systems such as cradle-to-farm gate LCAs or those involved in bioenergy products or services closely removed from feedstock production systems such as direct firing of wood pellets or chips. Given that LCA may be used to compare different end uses of biomass streams, the absence of an explicitly described functional unit makes these comparisons difficult to interpret.

ii. There is significant variation as to whether geographical attributes (spatial or temporal) are chosen to be included in functional units

There needs to be more consistency in the way that geographical attributes are applied to functional units. If the objective of the assessment is to focus on a particular product, then inclusion of geographical attributes in the functional unit should only be used if it is important to reflect impacts relative to spatial or temporal scales. Otherwise, geographical attributes may make comparisons between other LCAs difficult and make it harder to understand what the information being presented really means if the scales of the attributes are chosen arbitrarily or at inappropriate scales. It appears that practitioners are using space and time attributes in functional units as a means to identify what the resolution of impacts are relative to the larger product system, or at what scale aggregated data sets have been normalized or averaged across.

The spatial and temporal scales that are important for understanding environmental impacts of cultivation may be very different from the scales used to define the functional unit, and different still from the scales used to define the product system.

iii. LCI databases are typically average values aggregated from single farm or field measurements taken over large geographical areas

There needs to be transparency about the origin of underlying data that makes up life cycle inventories. Most existing commercial inventory databases have limited types of bioenergy feedstocks available for use; of those available, resolutions tend to be at national or multi-national scales. Although the data contained in many of the databases originate from individual farm or regions, the original spatial and temporal extent at which that data was collected is seemingly lost through aggregation. LCI data should be organized to clearly highlight spatial and temporal attributes of data; this could facilitate forward-thinking assessments that might better understand how local environments are functionally linked with agroecosystem activities and provide a better approach to understand land use change on a more local level.

iv. Inventory data from aggregated sources can be scaled a second time as impact assessments are calculated when geographical attributes are contained in functional units

There needs to be explicit details in LCA methodologies that discuss the effects of choosing to scale data compiled from secondary sources down or up to serve functional units. Inventory data used to represent feedstock data suggest that yields were derived from data averaged across large spatial and temporal scales. It was not previously understood that inventory data may be scaled a second time for impact assessments when geographical attributes are contained in functional units, but in completely inconsistent ways. Explicitly identifying geographical attributes of the data at each stage of the LCA would provide a simple way of being able to check the validity of some of these transformations and understand their comparability with other impact results.

v. Land use change is periodically discussed synonymously with changes in soil carbon

LUC needs to be discussed in terms of a collection of changes over time, and not focus on one aspect or indicator alone. LUC may involve changes to soil carbon, but there are a host of other impacts that make up those changes that can be ignored. Careful attention to how indicators and impacts are defined can help to avoid using terms that may reduce complex issues or results to single parameters of analysis.

3.6 Conclusions

This review highlighted the challenges one can encounter when attempting to interpret the environmental impacts of bioenergy technologies from studies that have not explicitly considered the role that geography has played in the data and results they generate. As more data from many different locations and scales of observations becomes available, without better direction on how to manage and integrate data from varying geographies in LCA, inconsistent approaches to system boundary selection and data management approaches will persist. The multi-scalar reality of bioenergy LCAs necessitates more explicit methodological direction in order to appropriately and consistently account for space-time differences implied by different scales of observation. Different information is generated at different research scales so the interpretation of data is scale-dependent and we need to be careful how we synthesize that data.

What is ultimately needed in LCA methods is a systematic way to start maintaining geographical attribute data so that systems and inventories are linked through space and time to start reducing the disconnect between earth systems processes and anthropogenic activities across shared space and time. This new approach would not be a radical change to LCA as it does not require fundamental changes to what practitioners are already doing. Instead it would advise those managing data through LCAs to maintain more rigorous geographical attribute background data, not for the purpose of addressing all dynamic challenges, but rather to create a more detailed database from which more robust tools and extensions could then use that information to create better connections with other areas of concern and facilitate better evaluations of systems that change in response to other system changes. Ultimately, improved transparency of geographical attributes across a number of LCA components is about realizing that if you know the system and its processes that you are describing better, you can make better interpretations from results.

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3.8 Appendix

Table A.3.1: Results of systematic review (2005-2015). All inputs listed with * are inferred and were not explicitly stated as such.

Reference	Feedstocks	Region	Scope	Functional Unit	Product/ Service	Spatial Extent (ha)	Yield Inputs (t_{dry}/ha/yr)	Yield Source	Temporal Extent Yield (yr)	Temporal Extent LCA (yr)	Soil C Accounting Method	Direct LUC Approach
Ashworth et al. (2015)	Switchgrass	U.S.	Cradle-to-Farm Gate	1 tonne dry weight switchgrass biomass	Feedstock Biomass	152	6.98 (5.68 - 9.83)	Farm Survey/ Measured Data	6	10	Not Identified/ Clear	Not Identified/ Clear
Bergman et al. (2015)	Switchgrass	U.S.	Cradle-to-Gate	1 Mg of switchgrass pellets (5% MC)	Feedstock Biomass	152	13.9	Farm Survey/ Measured Data	3	10	Not Identified/ Clear	Not Identified/ Clear
Daystar et al. (2015)	Switchgrass	U.S.	Cradle-to-Grave	1 MJ (HHV)	Transport Fuel Energy	Secondary Source - Not specified	Mass of feedstock delivered to plant per unit time	Not Identified/ Clear (GREET used for Inventory)	Secondary Source	1	Model/ Software	Direct LUC (modeled changes in soil carbon)
Hong et al. (2015)	Corn	China	Cradle-to-Gate	1 t ethanol production	Transport Fuel	National Averages (China)	Secondary Source	National & Regional Statistics (not explicitly stated)	Secondary Source	1	Not Identified/ Clear	Not Identified/ Clear
Iqbal et al. (2015)	Miscanthus, Switchgrass	Germany	Cradle-to-Farm Gate	1) 1 tDM 2) GJ/ha/yr	Feedstock Biomass, Energy produced per area of production	field trials (<1 ha)	miscanthus: 16.2, switchgrass: 10.2	Farm Survey/ Measured Data	11	1	Not Identified/ Clear	Not Identified/ Clear
Jeswani et al. (2015)	Miscanthus	U.K.	Cradle-to-Grave	1 litre ethanol	Transport Fuel	Secondary Source - Not specified	Secondary Source	GEMIS (Global Emission Model for Integrated Systems) (2004)	Secondary Source	1	Linear Emission/ Sequestratio n Factor	Direct LUC (modeled changes in soil carbon)
Meihui et al. (2015)	Corn	Taiwan	Cradle-to-Grave*	Not Explicitly Stated	Transport Fuel	Secondary Source - Not specified	6	Farm Survey/ Measured Data	4	1	Not Identified/ Clear	Not Identified/ Clear

Reference	Feedstocks	Region	Scope	Functional Unit	Product/ Service	Spatial Extent (ha)	Yield Inputs (t_{dry}/ha/yr)	Yield Source	Temporal Extent Yield (yr)	Temporal Extent LCA (yr)	Soil C Accounting Method	Direct LUC Approach
Murphy & Kendall (2015)	Switchgrass	U.S.	Cradle-to-Grave*	1 MJ of ethanol	Transport Fuel	Secondary Source - Not specified	5.7	Hrss et al. (2002), Jung & Lal (2011)	Secondary Source	1	Linear Emission/ Sequestration Factor	Direct LUC (modeled changes in soil carbon)
Parajuli et al. (2015)	Miscanthus	Denmark	Cradle-to-Grave*	1 MJ heat output	Heat Energy	Secondary Source - Not specified	30.23 (autumn harvest)	Uellendahl et al. (2008)	Secondary Source	1	Linear Emission/ Sequestration Factor	Not Identified/ Clear
Rugani et al. (2015)	Willow	Belgium	1) cradle-to-field gate 2) field gate-to-industrial gate	1) 1 m^3 of wood chips 2) 100 person km 3) 1 kWh 4) 1 GJ heat	Feedstock Biomass, Transport Distance, Heat Energy, Electrical Energy	1.03 - 1.74 (range over 5 sites)	9.755 (9.38 - 10.13)	Farm Survey/ Measured Data, Caslin et al. (2012)	Secondary Source	21	Model/ Software	Not Identified/ Clear
Schmidt et al. (2015)	Miscanthus, Switchgrass	Mediterranean	Cradle-to-Grave*	per 10 hectares in an average year	Area used for production	Secondary Source - Not specified	Miacanthus: 18 (fresh matter); Switchgrass: 10 (fresh matter)	Rettenmaier et al. (2010)	Secondary Source	1	Not Identified/ Clear	No LUC (similar or improved land/crop use assumed)
Sinistore et al. (2015)	Switchgrass	U.S.	Cradle-to-Gate	1 MJ (LHV of ethanol fuel)	Transport Fuel Energy	640,658	Secondary Source	Modeled (Environment Policy Integrated Climate (EPIC))	12	1	Model/ Software	Not Identified/ Clear
Wang et al. (2015)	Corn	U.S.	Well-to-Wheels	Not Explicitly Stated	Transport Fuel Energy	Secondary Source - Not specified	Not Identified/ Clear	Not Identified/ Clear	15	1	Not Identified/ Clear	Not Identified/ Clear
Yang & Suh (2015)	Corn	U.S.	Cradle-to-Gate	1) 1 ton crops 2) 1 ha/yr crop production	Feedstock Biomass, Area used for production	National Averages (U.S.)	Secondary Source	USDA-NASS (2000 - 2002)	3	1	Not Identified/ Clear	Not Identified/ Clear
Zaimes et al. (2015)	Miscanthus, Switchgrass	U.S.	Well-to-Wheels	1 MJ Renewable Fuel	Transport Fuel Energy	National Data	miscanthus: 15.29, switchgrass: 11.04	Farm Survey/ Measured Data, Secondary Sources (24 sources)	1 - 8 (depending on source)	miscanthus: 15, switchgrass: 10	Not Identified/ Clear	Not Identified/ Clear

Reference	Feedstocks	Region	Scope	Functional Unit	Product/ Service	Spatial Extent (ha)	Yield Inputs (t_{dry}/ha/yr)	Yield Source	Temporal Extent Yield (yr)	Temporal Extent LCA (yr)	Soil C Accounting Method	Direct LUC Approach
Caputo et al. (2014)	Willow	U.S.	Cradle-to-Farm Gate*	1 odt of willow biomass	Feedstock Biomass	Field trials in NY State	12.4 (10.7 - 14.1)	Volk et al. 2011	Secondary Source	21	Linear Emission/ Sequestration Factor	Not Identified/ Clear
Choudhary et al. (2014)	Switchgrass	U.S.	Cradle-to-Grave*	1) 1 MJ 2) 1 ha/yr 3) 1 km	Transport Fuel Energy, Transport Distance, Area used for production	Secondary Source - Not specified	12 (7-17)	Gunderson et al. 2008 (Davis et al. 2008, Groode 2008, Johnson et al. 2011)	Secondary Source	1	Not Identified/ Clear	Not Identified/ Clear
Garba et al. (2014)	Corn	U.S.	Cradle-to-Grave*	Not Explicitly Stated	Transport Fuel	Modelled - Not Clear	Secondary Source	Ecoinvent v2.0 database	Secondary Source	1	Not Identified/ Clear	Not Identified/ Clear
Hammar et al. (2014)	Willow	Sweden	Cradle-to-Grave*	1 ha arable land	Heat Energy	Secondary Source - Not specified	10 - 42 or 1st and then subsequent harvests	Ericsson et al. 2013, Hollsten et al. 2013, SOU 2007	Secondary Source	100	Model/ Software	Direct LUC (modeled changes in soil carbon)
Murphy et al. (2014)	Willow	Ireland	Cradle-to-Gate	1 GJ of energy in willow biomass	Feedstock Biomass	Secondary Source - Not specified	23, 30 for 1st &subsequent harvests respectively	Caslin 2010	Secondary Source	22	No Net Change Assumed	Not Identified/ Clear
Pieragostini et al. (2014)	Corn	Argentina	Cradle-to-Gate	1 MJ of anhydrous ethanol at biorefinery	Transport Fuel Energy	Secondary Source - Not specified	7.7	Ministerio de Agricultura, Ganaderia y Pesca, MAGyP (2010)	Secondary Source	1	No Net Change Assumed	No LUC (similar or improved land/crop use assumed)
Porso & Hansson (2014)	Willow	Sweden	Cradle-to-Gate*	1 GJ heat produced at a regional plant using willow or poplar pellets	Heat Energy	Secondary Source - Not specified	20, 30 for 1st and then subsequent harvests respectively	Not Identified/ Clear	Secondary Source	102	Model/ Software	No LUC (similar or improved land/crop use assumed)

Reference	Feedstocks	Region	Scope	Functional Unit	Product/ Service	Spatial Extent (ha)	Yield Inputs (t_{dry}/ha/yr)	Yield Source	Temporal Extent Yield (yr)	Temporal Extent LCA (yr)	Soil C Accounting Method	Direct LUC Approach
Sanscartier et al. (2014)	Miscanthus	Canada	1) cradle-to-gate* 2) cradle-to-grave*	1) 1 odt miscanthus bale 2) 1 odt pellet 3) 1 kWh	Feedstock Biomass, Electrical Energy	Secondary Source - Not specified	9.75 (8.4 - 11.1)	Deen et al. (2011), Vyn et al. (2012), Lewandowski & Heinz (2003)	Secondary Source	15-20	Model/ Software	Direct LUC (modeled changes in soil carbon)
Xue et al. (2014)	Corn	U.S.	Cradle-to-Farm Gate*	1 kg corn	Feedstock Biomass	U.S. Corn Belt	9.1 (6.9 - 11.3)	Not Identified/ Clear	Not Identified/ Clear	1	Not Identified/ Clear	Not Identified/ Clear
Buratti et al. (2013)	Corn	Italy	Cradle-to-Grave*	Not explicitly stated	Heat, Electricity	138	51.5 (49-54)	Farm Survey/ Measured Data	1	1	No Net Change Assumed	No LUC - Similar or improved land/crop use assumed
Geyer et al. (2013)	Corn, Switchgrass	U.S.	Well-to-Wheels	100 km driven /yr by compact passenger vehicle	Transport Distance	County Averages 150000 - 2389100 ha	Corn: 8 (4.4-11.6) Switchgrass : 13.9 (3.1-24.6)	USDA-NASS	5	1	Not Identified	Direct LUC - Area of Land Being Used
Godard et al. (2013)	Miscanthus	France	Cradle-to-Grave*	boiler heat production /yr (10000 MWh)	Heat	50km radius	Fresh Matter Value	Lin 2000, AGRESTE (2006)	20	1	AMG model/software	Not Identified/ Clear
Gonzalez-Garcia et al. (2013_a)	Corn	Italy	Cradle-to-Farm Gate	1 tonne biomass	Biomass Feedstock	2 farms in district of Lodi, 1 farm in district of Cremona	22.15 (15-29.3)	Farm Survey/ Measured Data	1	1	No Net Change Assumed	Not Identified/ Clear
Gonzalez-Garcia et al. (2013_b)	Willow	Sweden	Cradle-to-Grave	1) 1 km driven 2) 1 MJ heat	Transport Distance, Heat	*9058 (pool of commercial plantations)	4.8	Gonzalez-Garcia et al. (2012)	19	20	Not Identified/ Clear	Not Identified/ Clear
Han et al. (2013)	Corn	U.S.	Well-to-Wheels	1) 1 MJ fuel 2) 1 tonne biomass feedstock	Transport Fuel, Electricity, Bioproducts	97 km radius (farm to biorefinery distance)	Reported as energy per dry mass feedstock reported	No yields - /ton feedstock fuel yields (Jones et al. 2009)	Secondary Source	1	Not Identified/ Clear	Not Identified/ Clear
Murphy & Kendall (2013)	Corn	U.S.	Cradle-to-Farm Gate*	1 ha	Area of Production	National Averages (U.S.)	13.19	Not Accessible	Secondary Source	1	Not Identified/ Clear	Not Identified/ Clear

Reference	Feedstocks	Region	Scope	Functional Unit	Product/ Service	Spatial Extent (ha)	Yield Inputs (t_{dry}/ha/yr)	Yield Source	Temporal Extent Yield (yr)	Temporal Extent LCA (yr)	Soil C Accounting Method	Direct LUC Approach
Murphy et al. (2013)	Miscanthus	Ireland	Cradle-to-Farm Gate	1 GJ	Heat, Electricity	50 & 100 km transport distances (total cultivated area not detailed)	11.5	5 sources, 2 peer reviewed	Secondary Source	17	Not Identified/ Clear	Not Identified/ Clear
Yang & Chen (2013)	Corn	China	Well-to-wheels*	fuel needed for 1 km driven	Transport Distance	National Averages (China)	4.9	USDA-NASS	1	1	Linear Emission/ Sequestration Factor	Not Identified/ Clear
Budsberg et al. (2012)	Willow	U.S.	Cradle-to-Grave	1 MJ	Transport Fuel	WBP data - Not specified (80 km transport distance)	Total Mass of Feedstock produced per unit time	WBP (2011) - SUNY data	Secondary Source	21	No Net Change Assumed	No LUC - Similar or improved land/crop use assumed
Buonocore et al. (2012)	Willow	Sweden	Cradle-to-Grave*	Not explicitly stated	Heat, Electricity	100 ha (waste water filtered area - Lundby Gård)	Inputs per unit mass (kg) of feedstock or fuel produced	Farm Survey/ Measured Data	1	1	Not Identified/ Clear	Not Identified/ Clear
Dressler et al. (2012)	Corn	Germany	Cradle-to-Grave	1) 1 kg fresh matter 2) 1 kWh electricity	Heat, Electricity	13581	43.345 (39.8-46.9)	LSKN 2007, KFM project - links to a pdf	4	1	Not Identified/ Clear	Not Identified/ Clear
Goglio et al. (2012)	Corn	Italy	Cradle-to-Farm Gate	1) 1 ha 2) 1 GJ grain output yield	Biomass Feedstock	0.36	8.6	Bonari et al. 1992 - Suppl 1 - but not available and no link and title in italian	17	1	Not Identified/ Clear	Not Identified/ Clear
Gonzalez-Garcia et al. (2012_a)	Willow	Sweden	Cradle-to-Farm Gate	1 ha	Heat, Electricity	9048 (pool of commercial plantations)	5.29 (3.9-6.7)	Gonzalez-Garcia et al. 2012a, Borjesson & Tufvesson 2010	19	20	No Net Change Assumed	Not Identified/ Clear

Reference	Feedstocks	Region	Scope	Functional Unit	Product/ Service	Spatial Extent (ha)	Yield Inputs (t_{dry}/ha/yr)	Yield Source	Temporal Extent Yield (yr)	Temporal Extent LCA (yr)	Soil C Accounting Method	Direct LUC Approach
Gonzalez-Garcia et al. (2012_b)	Willow	Sweden	Cradle-to-Farm Gate	production of chips from 1 ha	Biomass Feedstock	9048 (pool of commercial plantations)	10.0	Mola-Yudego & Aronsson (2008) - available	19	20	No Net Change Assumed	Not Identified/ Clear
Pucker et al. (2012)	Miscanthus	Europe	Cradle-to-Grave	Not explicitly stated	Heat	Not Specified	30.0	Hofbauer et al. 2008 - not peer reviewed - need to translate	Secondary Source	1	Not Identified/ Clear	Not Identified/ Clear
Tonini et al. (2012_a)	Miscanthus Willow	Denmark	Cradle-to-Grave	1 ha Danish Agricultural land	Heat, Electricity	National Averages (Denmark)	Secondary Source	Hamelin et al. 2012	Secondary Source	20	Linear Emission/ Sequestration Factor	iLUC - Carbon emission factor
Tonini et al. (2012_b)	Miscanthus Willow	Denmark	Cradle-to-Grave	1 ha Danish Agricultural land	Heat, Electricity	National Averages (Denmark)	Secondary Source	not clear	Miscanthus: 21, Willow: 20	20	Model/ Software	iLUC - Carbon emission factor
Wang et al. (2012)	Corn, Miscanthus, Switchgrass	U.S.	Well-to-Wheels	1 MJ	Transport Fuel	National Averages (U.S.)	Inputs per unit mass (kg) of feedstock or fuel produced	Not clear what yields are	Secondary Source	Secondary Source	Model/ Software	Direct LUC - Modeled Carbon Changes
Yang et al. (2012)	Corn	U.S.	Cradle-to-Grave	fuel needed for 1 km driven	Transport Fuel	State Averages (U.S. - 19 states)	Secondary Source	N/A - given as area per bushel	5	1	No Net Change Assumed	Not Identified/ Clear
Blengini et al. (2011)	Corn, Miscanthus	Italy	Cradle-to-Grave	1 MJ of net energy delivered	Electricity, Heat	Not Specified	Corn: 70, Miscanthus: 22.5	LIFE, 2009; Monteleone, 2008	Corn: 1, Miscanthus: 18	1	Linear Emission/ Sequestration Factor	Not Identified/ Clear
Brandao et al. (2011)	Miscanthus, Willow	U.K.	Cradle-to-Farm Gate	1 ha/yr; later converted to 1 GJ to compare with energy systems	Biomass Feedstock	Average U.K. data	Miscanthus: 14.4, Willow: 9.5(9.5-20)	References 74, 89-92; 74 only one not peer review	Miscanthus: 20, Willow: 16	1	Linear Emission/ Sequestration Factor	Direct LUC - Land Use compared to alternate crops/scenario

Reference	Feedstocks	Region	Scope	Functional Unit	Product/ Service	Spatial Extent (ha)	Yield Inputs (t_{dry}/ha/yr)	Yield Source	Temporal Extent Yield (yr)	Temporal Extent LCA (yr)	Soil C Accounting Method	Direct LUC Approach
Eranki & Dale (2011)	Corn, Miscanthus, Switchgrass	U.S.	Cradle-to-Grave	5000 tons/day biorefinery (flows /kg feedstock)	Transport Fuel	280	Corn: Not Identified/ Clear, Miscanthus: 24.7 (19.8-29.6), Switchgrass: 21 (17.3-24.7)	Corn: WSS, Miscanthus: Gibson, 2008; Thelen et al. 2009, Switchgrass: Powlson et al. 2005, Godard 2007	Secondary Source	1	Not Identified/ Clear	Not Identified/ Clear
Fazio & Monti (2011)	Corn, Miscanthus, Switchgrass	Western Europe	Cradle-to-Grave	Not explicitly stated	Electricity, Heat, Transport Fuel	Western Europe Averages	Corn: 16, Miscanthus: 17, Switchgrass: 12	Two reports – not clear where to obtain	Secondary Source	1	Not Identified/ Clear	Not Identified/ Clear
Hussain et al. (2011)	Corn	Canada	Cradle-to-Grave	Not explicitly stated	Transport Fuel	Provincial Average	8.3	Data from Farm Tech Energy Corporation - no link or reference given	5	1	Not Identified/ Clear	Not Identified/ Clear
Kaliyan et al. (2011)	Corn	U.S.	Cradle-to-Grave*	1 MJ energy	Transport Fuel	Midwest Averages	9.6	Peer-reviewed Journals	10	1	No Net Change Assumed	Not Identified/ Clear
Kauffman et al. (2011)	Corn	U.S.	Cradle-to-Grave*	1 ha	Transport Fuel	Modelled yield for U.S.	11.6	EPA 2022 modelled yield; link not given	Secondary Source	30	Linear Emission/ Sequestration Factor	Not Identified/ Clear
Kimming et al. (2011)	Willow	Sweden	Cradle-to-Grave*	1 yr supply heat to 150 households	Heat, Electricity	250	8.0	Hansson et al. 2007 (peer reviewed)	Secondary Source	7	Linear Emission/ Sequestration Factor	Not Identified/ Clear
Whitman et al. (2011)	Corn	Canada	Cradle-to-Farm Gate	1 t dry corn stover delivered to EtOH plant	Biomass Feedstock	Total regional production	12.2	CRAAQ 2000 - Quebec publication	9	1	Linear Emission/ Sequestration Factor	iLUC - Carbon emission factor
Bai et al. (2010)	Switchgrass	not specified	Well-to-Wheels*	power to wheels for 1 km driving of a midsize car	Transport Energy	20, 40, 80 km transport distances	14.8	Bullard & Metcalfe 2001	20	20	Linear Emission/ Sequestration Factor	Not Identified/ Clear

Reference	Feedstocks	Region	Scope	Functional Unit	Product/ Service	Spatial Extent (ha)	Yield Inputs (t_{dry}/ha/yr)	Yield Source	Temporal Extent Yield (yr)	Temporal Extent LCA (yr)	Soil C Accounting Method	Direct LUC Approach
Cherubini & Jungmeier (2010)	Switchgrass	Austria	Cradle-to-Grave*	annual biomass treated (477 ktdry/yr)	Transport Fuel, Electricity, Heat, By-products	Not Specified	16.0	Smeets et al. 2009	Secondary Source	20	Linear Emission/ Sequestration Factor	iLUC - Carbon emission factor
Feng et al. 2010	Corn	U.S.	Cradle-to-Grave + Subsystem Scope	Not explicitly stated	Transport Fuel	51000 - 57900	9.38 (8.71-10.05)	Not Identified/ Clear	1	1	Linear Emission/ Sequestration Factor	Not Identified/ Clear
Hoefnagels et al. (2010)	Corn, Miscanthus Switchgrass	Varied	Cradle to Grave	1) MJprim/MJfuel, 2) MJprim/ha	Heat, Electricity	Average Value (/ha)	Not Identified/ Clear	not clear	Secondary Source	1	Linear Emission/ Sequestration Factor	Direct LUC - Carbon emission factor
Hsu et al. (2010)	Corn, Switchgrass	U.S.	Field-to-wheels	1 km travelled by a light-duty passenger car on E85 in 2022	Transport Distance	Modelled - Not Clear	Secondary Source	Peer-reviewed journal	Secondary Source	1	Not Identified/ Clear	Not Identified/ Clear
Schumacher et al. (2010)	Corn	Germany	Cradle-to-Grave*	energy yield /ha/yr	Transport Fuel	1000	20.0	Schumacher 2008 - German dissertation	1	1	Not Identified/ Clear	Not Identified/ Clear
Stephenson et al. (2010)	Willow	U.K.	Cradle-to-Grave*	1 tonne bioethanol	Transport Fuel	100000	10 (5.0-15)	not clear	Secondary Source	1	Model/ Software	Direct LUC - Modeled Carbon Changes
Goglioa & Owende (2009)	Willow	Ireland	Cradle-to-Grave*	1) ha, 2) energy output/input ratio 3) CO_2emission/ unit net electrical energy	Electricity	15	10 (8-14)	McCracken, (2007) (book), Caslin (2008)	25	25	Not Identified/ Clear	Not Identified/ Clear
Hillier et al. (2009)	Miscanthus	U.K.	Cradle-to-Farm Gate*	Not explicitly stated	Biomass Feedstock	Not Specified	9 (5-18)	Richter et al. (2008)	1	1	Model/ Software	Direct LUC - Land Use compared to alternate crops/scenario

Reference	Feedstocks	Region	Scope	Functional Unit	Product/ Service	Spatial Extent (ha)	Yield Inputs (t_{dry}/ha/yr)	Yield Source	Temporal Extent Yield (yr)	Temporal Extent LCA (yr)	Soil C Accounting Method	Direct LUC Approach
Kim et al. (2009)	Corn	U.S.	Cradle-to-Farm Gate	1 kg dry biomass (grain or stover)	Biomass Feedstock	County Level Data	7.49 (5.9-9.1)	USDA NASS (2007)	4	1	Model/ Software	No LUC - Similar or improved land/crop use assumed
Liska et al. (2009)	Corn	U.S.	Cradle-to-Grave*	Not explicitly stated	Transport Fuel	State Averages	Not Identified/ Clear	USDA-NASS	Not clear	1	No Net Change Assumed	Not Identified/ Clear
Monti et al. (2009)	Miscanthus Switchgrass	Western Europe	Cradle-to-Farm Gate	1) 1 hectare 2) 1 GJ	Biomass Feedstock	Not Specified	Miscanthus: 17, Switchgrass: 12	Venturi & Venturi 2003	Secondary Source	1	Not Identified/ Clear	Not Identified/ Clear
Yu & Tao (2009)	Corn	China	Cradle-to-Grave	Not explicitly stated	Transport Fuel	Not Specified	Not Identified/ Clear	N/A - given as per ton feedstock	1	1	Not Identified/ Clear	Not Identified/ Clear
Kim & Dale (2008)	Corn	U.S.	Cradle to Grave*	1 kg of ethanol	Transport Fuel	Not Specified	Inputs per unit mass (kg) of feedstock or fuel produced	USDA-NASS 2005	4	1	Model/Software	Not Identified/ Clear
Pelletier et al. (2008)	Corn	Canada	Cradle-to-Farm Gate	1 kg each crop produced, at the farm gate	Biomass Feedstock	* State Level	7 (6.8-7.2)	Badgely et al. (2007)	Secondary Source	1	Not Identified/ Clear	No LUC - Similar or improved land/crop use assumed
Renouf et al. (2008)	Corn	U.S.	Cradle-to-Plant Gate	1 kg monosaccharide (glucose or fructose)	Transport fuel	State Averages	9.1 (7.8-9.5)	Kim and Dale (2004)	Secondary Source	1	Not Identified/ Clear	Not Identified/ Clear
Searcy & Flynn (2008)	Corn	U.S.	Cradle-to-Grave	1) /Mwh, 2) /Lfuel, 3) /L ethnol	Electricity, Transport Fuel	Regional Statistics - Not Specified	Secondary Source	Peer-Reviewed Journals	Secondary Source	1	No Net Change Assumed	Not Identified/ Clear
Adler et al. (2007)	Switchgrass	U.S.	Cradle-to-Grave	Not explicitly stated	Electricity, Transport Fuel	County Level Data	Inputs per unit mass (kg) of feedstock or fuel produced	USDA-NASS 2004	10	1	Model/Software	Direct LUC - Area of Land Being Used

Reference	Feedstocks	Region	Scope	Functional Unit	Product/ Service	Spatial Extent (ha)	Yield Inputs (t_{dry}/ha/yr)	Yield Source	Temporal Extent Yield (yr)	Temporal Extent LCA (yr)	Soil C Accounting Method	Direct LUC Approach
Styles & Jones (2007)	Miscanthus, Willow	Ireland	Cradle-to-Farm Gate	Not explicitly stated	Biomass Feedstock, Electricity	Miscanthus: 34055, Willow: 39899	Miscanthus: 11.7 Willow: 8.8	Gigler et al. 1999 (willow), Clifton-Brown et al. 2000 (misc)	Secondary Source	Miscanthus: 16, Willow: 23	Linear Emission/ Sequestration Factor	Direct LUC - Land Use compared to alternate crops/scenario
Wang et al. (2007)	Corn	China	Cradle-to-Farm Gate	Not explicitly stated	Transport Fuel	National Data	7.7	USDA stats	1	1	Not Identified/ Clear	Not Identified/ Clear
Kim & Dale (2005_a)	Corn	U.S.	Cradle-to-Grave	1) 1 kg ethanol 2) 1 km driven by midsized passenger car	Transport Fuel	County Level Data	Inputs per unit mass (kg) of feedstock or fuel produced	USDA stats	3	1	Model/ Software	Not Identified/ Clear
Kim & Dale (2005_b)	Corn	U.S.	Cradle-to-Grave	1 ha for 40 yr period for each crop type	Transport Fuel, Biomass Feedstock, Electricity	County Level Data	Inputs per unit mass (kg) of feedstock or fuel produced	local county data from US EPA model	4	40	Model/ Software	Not Identified/ Clear
Spatari et al. (2005)	Corn, Switchgrass	Canada	Well-to-Tank	Not explicitly stated	amount of fuel	Not Specified	Corn: Secondary Source, Switchgrass: 8 (6-13)	not clear (sensitivity analysis used for ranges)	Secondary Source	1	No Net Change Assumed	Not Identified/ Clear

Chapter 4 – Life Cycle Analysis of Willow Biomass Feedstock in southwestern Ontario: Demonstration of a Multi-Scale Modular Structuring Approach

Abstract

In Ontario and other regions of Canada, there is interest in expanding the use of bioenergy as part of emerging renewable energy portfolios and sustainable land-use and planning approaches. There have been conflicting LCA results between studies when bioenergy systems are considered and this has created doubt about the long-term sustainability of bioenergy and limited expansion of its use. Many of these issues have been attributed to site-specific geographical differences or gaps in LCA standard methods; however, there remain significant challenges when LCA is used to evaluate multi-scale environmental systems, such as those that include biomass feedstock production. In this chapter, a modular multi-scale approach to structuring goal and scope definitions, and their link to inventory and impact assessment phases, are evaluated in the context of performing a cradle-to-farm gate LCA. Short-rotation coppice (SRC) willow grown in southwestern Ontario (Guelph) as a bioenergy feedstock is considered for a 21-year period using a combination of field and modeled data, in order to understand net energy use and total greenhouse gas (GHG) emissions. A modular biomass production and soil carbon model is assessed and compared with traditional aggregated emission factor approaches. Results show that improving the spatial and temporal resolution of input data can improve site-specific decision-making that high-level LCA approaches cannot address. The approach used in this study also shows that improving resolution of underlying attributes associated with data does not eliminate the ability to continue to conduct existing LCAs, suggesting that taking such modular multi-scale approaches can improve site-specific LCAs in cooperation with existing approaches.

4.1 Introduction

Bioenergy is an emerging renewable energy source being explored in Ontario that has the potential to maximize biomass resources by providing new fuel and energy products that can complement existing forestry, food, and fibre industries. One feedstock gaining interest as a potential bioenergy source in Ontario is short-rotation coppice (SRC) shrub willow (*Salix spp.*) that can be grown on marginal or degraded land and therefore minimize indirect land-use change (iLUC) effects (Zamora et al. 2014; McElroy & Dawson 1986; Amichev et al. 2012). Determining appropriate locations to deploy dedicated bioenergy feedstocks such as SRC willow requires an assessment of their potential benefits and environmental impacts compared to other land-use options. Accurately identifying environmental impacts is of critical importance in order to provide downstream customers and policy decision-makers with information about the long-term competitive advantages of willow over other land-use alternatives; some of these alternatives may also include other competing renewable energy options (Calvert & Mabee 2015).

As illustrated in previous chapters of this thesis, LCA approaches have tended to account for farm scale processes and impacts by using average values or emission factors derived from large aggregated datasets; these serve to broaden its application and accessibility but have limited utility to describe finer-scale changes of on-farm dynamics. Gaps in understanding ecosystem impacts at fine scales may lead to misevaluation of those processes when they are assessed at larger scales; this may be the case when trying to understand factors like soil carbon dynamics, and the relationship that crop residues and management at the field-scale may have on both. In order to evaluate bioenergy systems at the farm or landscape perspective, better direction in setting the terms of LCA is required, and better options for organizing LCA are needed so that they can be more adaptable to changing management practices, variable environmental conditions, and the dynamics of nutrient cycling such as carbon and nitrogen in soil. The challenge lies in integrating processes and activities that occur at different spatial and temporal scales within a common operational framework: soil processes, biomass growth, and land management activities are all best observed at different spatial and temporal scales and may require multiple perspectives to effectively understand each effectively.

One solution to managing system dynamics where processes are operating at multiple scales is to embrace variability, and to approach LCA with a more flexible modular approach. Adoption of modular system theory has proved useful in examining issues for a number of multi-scale systems including management of complex systems (Campagnolo & Camuffo 2010), production line manufacturing (Yang et al. 2013), understanding market uncertainties (Xu et al. 2012), and more recent cloud-based management approaches (Mital et al. 2015). Schilling (2000) has pointed out that system heterogeneity tends to increase as geographies change and systems become more complex. A modular framework can be designed to handle data at different spatial and/or temporal scales; in this case, modularity defines the degree to which different system components are coupled to each other. A framework that can manage and integrate processes across a range of scales simultaneously is of particular interest given the multi-scalar reality of different LCA components.

As identified in previous chapters, LCAs are in many ways already informally modulated as the spatial and temporal scales at which product systems, process boundaries, and functional units are characterized are often different. The ways in which these different scales are managed in the context of LCA objectives and standards are inconsistent across the literature. For example, in a review of LCA studies that evaluated ethanol production, soil carbon emissions and balances that were included in their impact assessments were accounted for using three different methods that used data at varying spatial and temporal resolutions (Manion, 2017). If the goal is to enable LCA with more accurate representations of the processes at work, it needs to be capable of considering spatially- and temporally-discreet information and attributes across a wide range of human and environmental observations. Unfortunately, no formal framework exists which can facilitate the management of geographical attributes, and which can effectively and transparently describe these multi-scale systems. A useful data management framework would be one that can explicitly maintain spatial and temporal attributes of system, inventory, and impact data in a transparent and accessible database.

4.1.1 Study Objective

Most LCAs related to bioenergy and biofuels define their product systems at a single particular geographic scale, and proceed to structure inventory analysis and impact assessments based on a singular functional unit (FU) that broadly groups life cycle stages, groups of processes, management influences, or unit processes based on the objective of the LCA. These studies rarely utilize more than one spatial or temporal scale of analysis across each of the LCA stages. In this study, a modular LCA method that takes focuses on several different spatial and temporal scales within the feedstock production stage (on-farm) of a SRC willow plantations is proposed and tested. The aim is to explore the differences in results that such an approach provides compared to more traditional LCA methods that rely on structuring approaches with minimal differences and often singular scales of observation. An important concept in modular system design is that the different subsystems can be substituted without a significant change to the overall system structure or functionality of the system (Schilling 2000). Rather than compare results to a reference system or different end products, the approach of this LCA will be to evaluate the same system using multiple scales, representing different process and data resolutions.

To facilitate the comparison of the approaches described above, a LCA of the environmental impacts of the production of SRC willow in southwestern Ontario is performed. The modular approach taken to evaluating willow production is based on two current realities: first, that there are some areas and compartments of production that may be assessed at different scales than others; and second, that for any given component or area the level of detail available for that data may vary from one location to the next. The LCA is organized in a modular fashion, with each module being intended to account for the flow (inputs and outputs) of variables in discrete components of the agroecosystem based on a chosen FU. Model performances are validated with direct values measured in Guelph, Ontario along with previously published literature data. The expectation is that using a modular approach will facilitate the addition of new models or more detailed versions of existing pathways to allow for the simultaneous modeling of effects associated with changing environmental impacts. By separating processes and activities occurring at different spatial and/or temporal scales, the aim is to allow for simultaneous evaluation of impacts and technological change across a range of data with varying quality, which has been an ongoing issue with the static approaches of traditional LCAs. This approach has the potential to provide a better understanding of GHG dynamics and could provide optimal solutions to maximize emissions reductions.

4.2 Methods

4.2.1 Goal & Scope

The LCA goal is to evaluate the potential environmental impacts of producing SRC willow biomass on former cropland in Guelph, Ontario, following ISO 14044:2006 standards. The plantation is assumed to operate for 21-years and the impact indicators evaluated are GHG emissions (kg CO_2eq), net energy use (GJ), and change in soil carbon (kg C). Annual field data used to inform land management inputs were obtained from nearly 1 ha (0.67 ha) of SRC willow established in Guelph between 2006 and 2009, and so the main functional unit chosen to represent results was one oven-dried tonne of biomass per hectare per year (odt $ha^{-1}yr^{-1}$).

The extent of the supply chain assessed was defined as cradle-to-farm gate (Figure 4.1) that can be broadly structured into three different groups of processes: upstream product and energy production, upstream product transport, and biomass production. Most traditional LCAs would compare each of the three process groups in a similar manner, using large-scale aggregated emission factors and crop yield results, observed elsewhere, that largely ignore site-specific differences. The traditional approach is described in the flow diagram shown in Figure 4.1: environmental flows and soil processes are not described in any sort of detail. In order to evaluate site-specific crop and soil dynamics, this study divides the biomass production stage into more specific unit processes as described in the next section.

4.2.1.1 Biomass Production Boundaries

The location chosen for production is Guelph, Ontario, Canada at the Guelph Agroforestry Research Station (43°32'28" N, 80°12'32" W). The site was prepared and willow cuttings planted in May 2006. The crop was coppiced in after its first season and its first rotation harvest was in December of 2009. Plots (9.25m x 50 m) of each of three willow clones – *Salix dasyclados* (SV1), *Salix miyabeana* (SX67), and *Salix purpurea* (9882-41) – were obtained from the State University of New York's College of Environmental Science and Forestry (SUNY-ESF) and arranged randomly within each field for four replications, and a total of 12 plots in each field. Willow plots consist of five double rows, with alternating inter-row distances of 0.75 m and 1.5 m, and spacing between cuttings is 0.55 m, resulting in a planting density of approximately 20,000 stems ha^{-1}. Fuel use, energy consumption, and chemical (fertilizer and herbicide) applications were recorded directly from cultivation of the SRC willow plantation established at the site plots.

In this study, four different modules used to organize inventory analysis and impact assessments that utilize different structuring approaches and scales of analysis: land management, crop growth, soil carbon, and soil nitrogen (Figure 4.2). For the land management module, the processes considered follow the traditional flow shown in Figure 4.1 where direct and indirect emissions from fuel and products used on site are compiled using generalized emission factors and only assesses impacts at once scale across all scenarios. The flows considered, the spatial and temporal scales of those flows, and number of scenarios that each of the remaining three modules assess are described in detail in the life cycle inventory (LCI) analysis phase below.

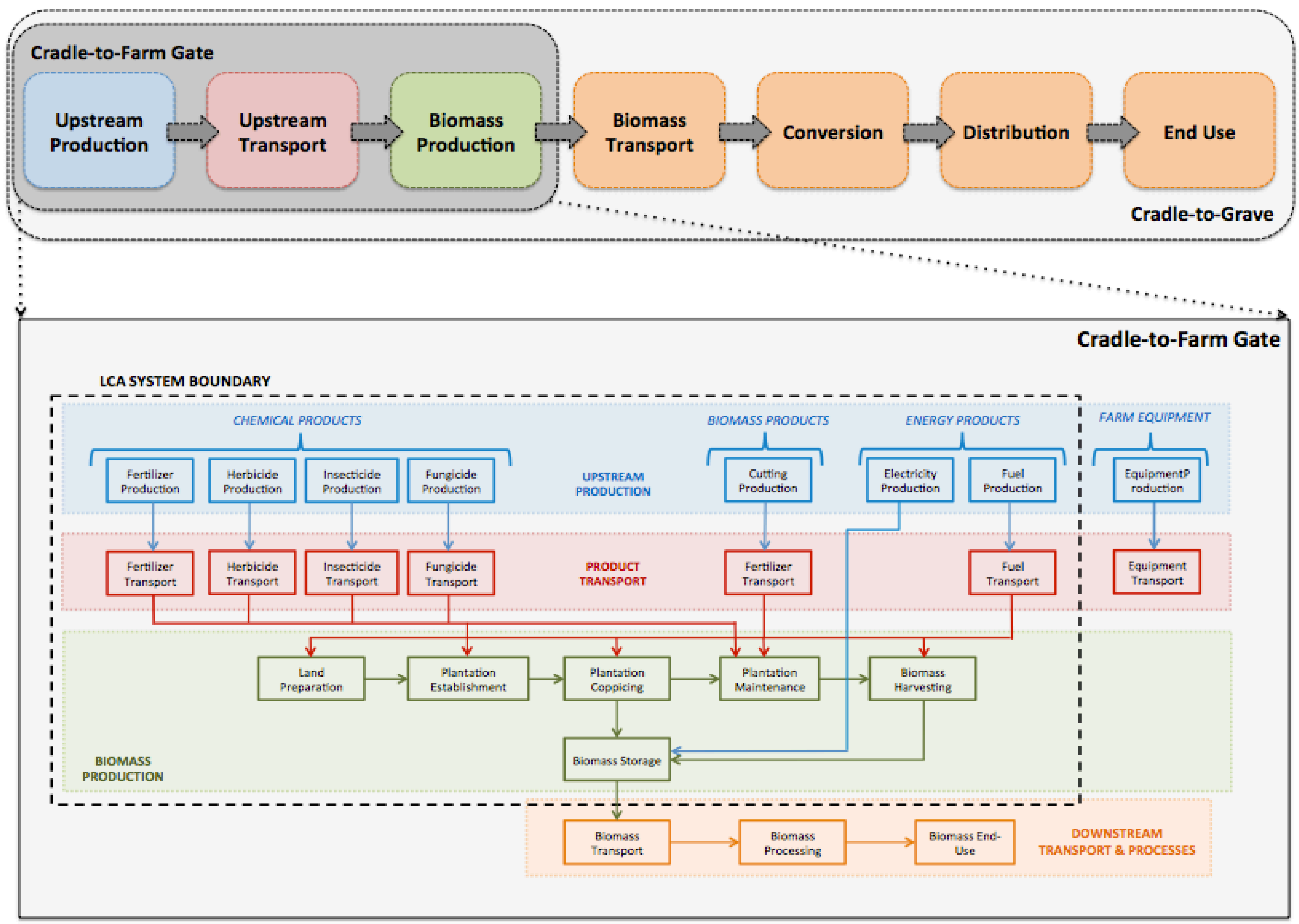

Figure 4.1. Typical LCA System Boundary that includes upstream production (chemical products, biomass products, energy products), product transport from production to the farm, and on-farm processes associated feedstock production.

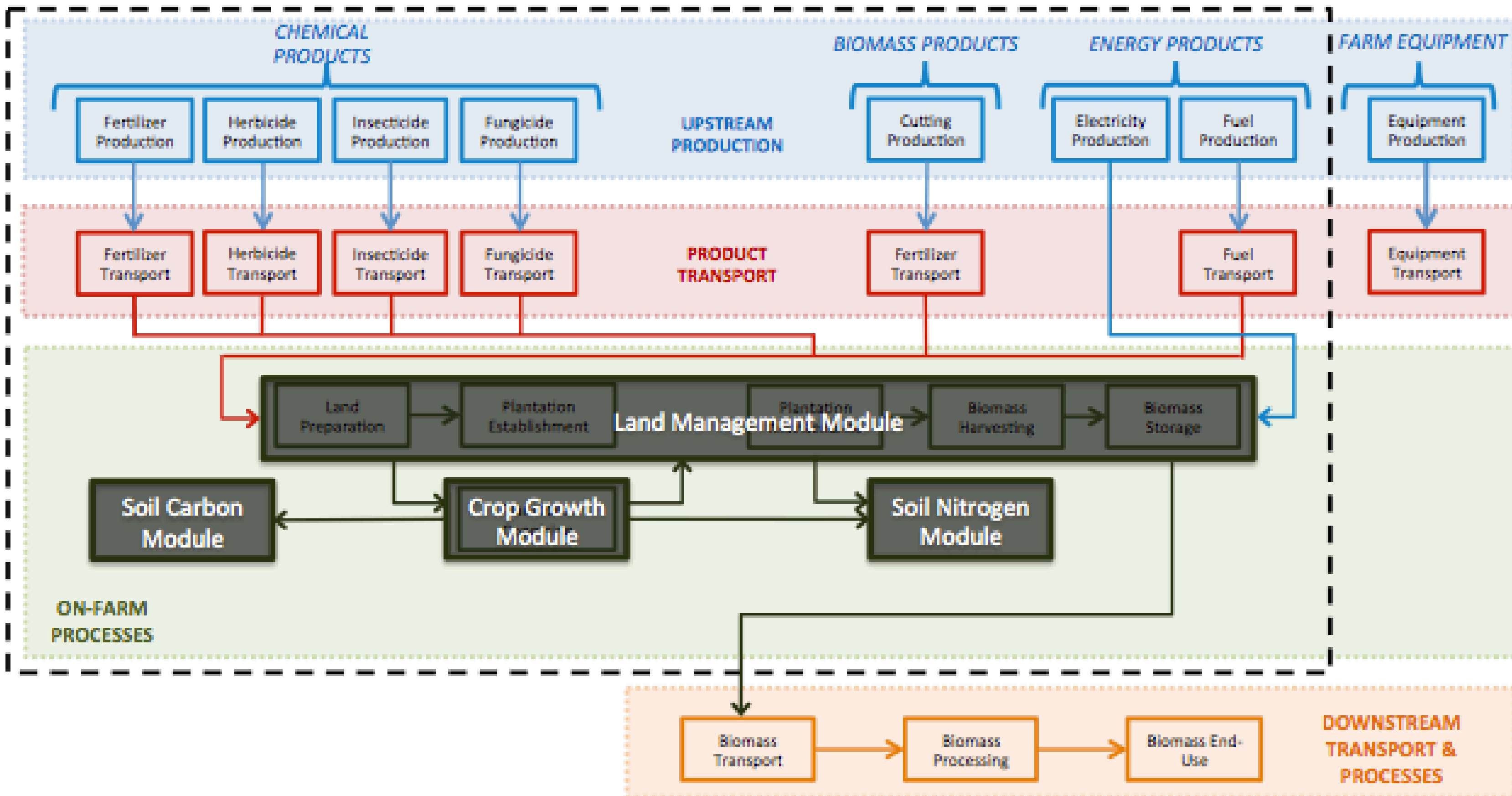

Figure 4.2. Modular components of the biomass production phase of the LCA. The Land Management Module contains the flows for the biomass production stage shown in Figure 4.1, but is now linked with additional components of the biomass production system. Decisions made in the Land Management Module directly affect inputs to soil nitrogen from fertilizers and when crops are harvested, while residues calculated from the crop growth module directly influence carbon and nitrogen inputs to the soil modules.

Table 4.1. Comparison of the modular approaches compared with the conventional approach to evaluating each module within the modular framework. Only the soil carbon module is evaluated at a second resolution.

Module	Conventional LCA	Modular LCA – Resolution #1	Modular LCA – Resolution #2
Soil Carbon	IPCC Tier I Approach – No Net Change Assumed	Linear Sequestration Factor – Annual factor	Dynamic Model – Daily time-step
Soil Nitrogen	IPCC Tier I Approach – Global Emission Factor	IPCC Tier II Approach – Ontario Emission Factor	N/A
Crop Growth	Average Annual Yield	Variable Annual Yield	N/A

4.2.2 Life Cycle Inventory (LCI)

4.2.2.1 On-Farm Product & Energy Use

The total amount of fuel (diesel) and chemical products (fertilizers, herbicides) assumed to be used at the farm scale during the entire 21-year life cycle of willow production are shown in Table 4.1 on a per hectare basis. Estimates of fuel and chemical use rates were based on direct measurements from in-field activities performed at the Guelph agroforestry site between May 2006 and December 2009. Operations performed between 2007 and 2009 included all activities associated with a regular rotation harvest cycle and so the values recorded during that period are assumed to be average values for each subsequent rotation over the 22-year life-span. Fertilizer application was performed in the spring following harvest years and weeding between harvests was performed manually or using mechanical disking or rototilling. Willow harvests and coppicing cut the willow at 2-4 cm height using a sickle bar mower.

Table 4.2. Products and fuel used for site preparation, establishment, operations, and decommissioning over a 21-year period, reported on a per hectare basis. All amounts are based on three years of field trial data scaled up to 21 years.

Parameter	Unit	Amount
Site Preparation & Establishment		
Planting density	Cuttings/ha	20,000
Dual Ii Magnum (S-Metacholr)	kg a.i./ha	0.92
Goal 2XLm (Oxyflurofen)	kg a.i./ha	1.12
Urea	kg/ha	75
Triple Super Phosphate (TSP)	kg/ha	25
Muriate of potash(MOP)	kg/ha	50
Fuel use (preparation, planting)	L/ha	44.19
Operations, Decommission		
Urea	kg/ha	611
Triple Super Phosphate (TSP)	kg/ha	244
Muriate of potash(MOP)	kg/ha	395
Glyphosate for decommissioning	Kg/ha	2.5
Fuel use (maintenance, harvest, loading)	L/ha	180
Fuel use for decommissioning	L/ha	1.11

4.2.2.2 Product Transportation

The distance involved in transporting products and fuel used on-farm, and the fuel efficiency of the transportation methods, are listed in Table 4.2. All transportation emissions are calculated based on these factors, using diesel as the fuel type and using reference data from Environment Canada (2015) to determine the fuel efficiency of each transportation method. Transport distances listed are for return trips; the amount of fuel used and the distance transported represent twice that needed for a one-way trip.

4.2.2.3 Upstream Product & Energy Emission Factors

For all fuel and products used during farm activities, impacts from production and energy use upstream are calculated using emission factors as shown in Table 4.3. Upstream data for chemical production was originally taken from Nagy (2000) and updated in 2006, while all fuel and transport emission factors are derived from Environment Canada (2016) emission factors where available. Any remaining factors were generated from IPCC values (Table 4.3). Emission factors were used to calculate GHG emissions and net energy used for all upstream product, energy, and fuel use inventories; the calculations used for each can be found in Appendix 4.1. For upstream emissions associated with the production of willow cuttings, data was made available through the University of Guelph through correspondence with Agro Energie Willow Nursery, Montreal, QC.

Table 4.3. Transport distances of chemicals used and willow cuttings and the fuel use efficiencies of the transport methods for each. Transport distances represent a round-trip between the farm and the supply location for each product transported.

Product	Distance (round-trip, km)	Transport Method	Efficiency (L/100km)	Source
Fertilizers & Herbicide	160	Pick-up Truck	20	Environment Canada 2015
Willow Cuttings (Montreal to Guelph)	1236	Transport Truck	40	Environment Canada 2015

Table 4.4. Greenhouse gas and energy use emission factors (EF) for upstream product and fuel production emissions.

Parameter	GHG (kg CO_2eq)	Net Energy (MJ)	EF units	Source
Urea (46% N)	4.099	70.88	/kg N	Nagy 2000
TSP (55% P_2O_5)	0.5699	9.53	/kg	Nagy 2000
Potash (K_2O)	0.2443	6.99	/kg	Nagy 2000
Herbicide (High)*	699.5	10.4	/each	Nagy 2000
Willow Cuttings I	0.107	1.328	/cutting	Agro Energie Willow Nursery, Montreal, QC
Diesel Production	2.992	35.8	/L diesel	Environment Canada, 2015

*The herbicide emission factor chosen was the product with the largest greenhouse gas emissions (Achieve X Gold) to maintain a conservative approach.

4.2.2.4 Biomass Production Module

4.2.2.4.1 *Average Annual Yields*

Previous LCAs that included willow biomass production in their system boundaries often inventory willow yields by estimating the total harvestable yield for the entire life cycle of the plantation and allocating that value to each year of operation, obtaining an average annual yield (e.g. odt/ha/yr). Annual yields used have generally been based on observations from short-term field trials (<5 years) performed on or near the land being assessed (see Heller et al. 2003; Kumundinie et al. 2012), or by using long-term field trials (>10 years), which can also be obtained on-site but often consists of data aggregated from larger spatial areas or from a number of references (see Gonzalez-Garcia et al. 2013b, 2012a, 2012b; Brandao et al. 2011; Goglio &

Owende 2009). For the baseline scenario of this LCA, the annual yield approach to calculating willow biomass harvests is approached using a previously used conservative annual average of 7 odt/ha/yr for southwestern Ontario (Kumundinie et al. 2012) based on short-term (2006-2009) field trials conducted at the Guelph Turfgrass Institute.

4.2.2.4.2 Variable Yield Estimates

Actual willow growth is likely to vary on an annual basis and so there is a need to estimate those variations. From a practical perspective, fluctuations in annual yields can have implications for downstream use of biomass feedstocks, affecting both pricing and supply of energy. From an impact assessment perspective, variations in the way yield is estimated affect the rate at which carbon is sequestered which could affect long-term results, and perhaps more critically biomass yields are often directly linked to other components involved in feedstock production, and so creating static yields invariably results in static inputs to those other components. Crop residues are directly related to yields and are the single largest input of annual organic material to agroecosystem soils; therefore, understanding annual changes in yields is critical in being able to understand finer temporal scale changes in other components of feedstock production.

There are challenges in finding models appropriate to estimate yield variations as they are not yet widely available. Recent work (Wang et al. 2015) has successfully modeled willow growth in response to fine scale (hourly) environmental changes, but it has only been applied to a single 4-year rotation and has been not integrated with land management activities to understand how changes in growth occur in response to coppicing and different crop management options. The other limitation of finer-scale models is that there is currently limited data available to operate such a model at the site being evaluated, which is a well-established pre-existing issue with bioenergy LCAs. At annual intervals, there is some consensus on general patterns of growth that might be appropriate to use in order to estimate annual variation. In SRC willow growth, plantations that have been coppiced at the end of the first year tend to increase in annual yield until a site-specific maximum yield is attained, usually in Years 5 or 6, which are the first years following the first rotation harvest (Kopp et al. 2001). Annual yields in the years after maximum yields are attained and up until the plantation life cycle ends tend to fluctuate around the maximum yield, but can vary annually depending on seasonal changes in temperature or precipitation, and on site-specific environmental and land management factors.

Yield data from the field trials conducted in Guelph from 2006 – 2009 are available for the first three years of growth following establishment. Yields for years 4 to 21 were not measured and need to be estimated in order for annual variation for the entire life cycle to be considered. To account for annual variation in yield from the point of site establishment until a maximum yield is obtained, yields were estimated for years with missing yield data, and for conducting sensitivity analyses of yields, using the logistic curve equation presented in Kopp et al. (2001). In the 16-year study at NY State SUNY-ESF (Kopp et al. 2001), 10 annual yields from willow trials were found to increase in a non-linear (logistic curve) pattern between crop establishment and willow maturity (around year 5) when a maximum obtainable yield was observed. The growth pattern was best described using the equation:

$$Y=A/(1+Be^{nx}) \qquad [1]$$

where Y is the annual biomass production (odt/ha/yr), x is the age of the plantation (years), n is a value that reflects the intrinsic rate of increase, and B is a constant. Values for n (-2.4) and B (250) were calculated from an average of each of 12 trials for seven different willow cultivars in

Kopp et al. (2001). The final parameter A represents the maximum observable yield that willow is likely to attain.

The maximum obtainable yield estimated to use as the baseline value for this LCA is 9.25 odt/ha/yr, based on yields observed from the first rotation harvested at Guelph in 2009 (9.1 odt/ha). Maximum yields are expected to occur in the years following the first rotation so a small conservative increase in yields compared to the 2009 harvests was used. Using the maximum observable yield of 9.25 odt/ha/yr, default values for *B* and *n* were used for the curve as no adjustment of the values during calibration was found to improve the curve. The observed and predicted annual yields for 2006-2009 are shown in Table 4.4 and a regression analysis between the data sets were found to fit reasonably well (R^2=0.9228, $p<0.05$). While years 2 and 3 are not as accurate as years 1 and 4, the precipitation in year 2 (474.5 mm) following the first year of coppice was significantly below the historical (1981-2010) average (776.8 mm) (Environment Canada, 2016). Results of the regression suggest that the equation is valid to use for estimating the yield for years 5 and 6, and for years 0 – 6 when the maximum yield is changed for sensitivity analyses.

In the study by Kopp et al. (2001), willow yields were found to fluctuate between 60 and 87% of the maximum attainable yield for years 6 to 10. To provide an estimate of willow yields for years 7 to 21 in this LCA, a random value between 60 and 100% of the maximum obtainable yield is used for years 7 to 15 to account for annual weather variations, and a random value between 60 and 90% of the maximum obtainable yield is used for years 16 to 21. The percent of the maximum yield selected for each year and the corresponding yield used to test the sensitivity of LCA results and the soil carbon model are available in Appendix 4.2. The percent of the maximum yield selected for each year is applied the same for each change in maximum yield during sensitivity analysis to maintain consistency.

Table 4.5. Observed versus Predicted Yields for the first 4 years of willow production at Guelph Ontario. Regression analysis of the two yield groups resulted in a good fit of data with R^2=0.9228 ($p<0.05$) and y=1.1433x – 1E-14

Year	Predicted Yield (odt ha^{-1} yr^{-1})	Average Observed Yield (odt ha^{-1} yr^{-1})
2006	0.384	0.54
2007	2.976	1.11
2008	7.669	4.5
2009	8.948	9.1

4.2.2.4.3 Biomass Partitioning

Determining the mass of each biomass fraction is not straightforward due to variations in the way that willow allocates biomass growth to each of the three major structures (leaves, stems, and roots) in response to environmental, land management, and age-related influences. For example, previous work has identified how fine root growth and depth can be inhibited by fertilizer applications (Heinsoo et al. 2009) and promoted by moisture gradients in soil profiles (Stadnyk 2010). Shoot, root, and leaf growth are also known to vary within growing seasons where roots may continue growing in early spring or fall when shoot growth has ceased (Rytter & Hansson 1996), and where leaf growth can dominate net primary production allocation during leaf-out (Cannell 1988; Lindroth 1994). Biomass partitioning has also been observed to vary with the age of the plantation where root:shoot ratios tend to be largest (0.5 – 2) in the early years of willow plantations until coarse roots reach a steady mass at which point increased aboveground production may reduce those ratios (Pacaldo et al. 2012). While partitioning biomass production based on responses to environmental, land management, and age-related factors could produce a highly accurate picture of residue production, the uncertainty of how those factors actually affect those responses in current literature is not well understood. There are also significant gaps in data available for the site in question, which limits the ability to utilize links between factors - even if they were widely understood. As a result, biomass partitioning into leaf, coarse root, and fine root fractions are based on allocation approaches that take average annual production and allocate it into daily time intervals

While yields are based on annual averages, the overall LCA model accounts for radiation, temperature, soil moisture, residue inputs, and soil carbon decomposition at daily time intervals. To integrate the yield estimates with overall model functions at daily time intervals, the annual yield is averaged on a daily basis over the growing period. The growing period for willow in this LCA is assumed to begin on May 1 (Day 121) and end on October 15 (Day 288). Leaf and root growth are assumed to occur during the same period and allocation to those biomass fractions are discussed in the sections below.

To determine the mass of litterfall produced, a fixed ratio of leaf to shoot production is assumed for each year of production based on litterfall collections performed at the Guelph site in 2009. At the end of a three year rotation, 1673 kg ha^{-1} $yr^{-1}$of leaf biomass was recorded between June and November of 2009 (Cardinael et al. 2012; Clinch 2009). When leaf biomass is compared with the shoot yields obtained in 2009, the leaf:shoot ratio was 0.554. As a result, the total annual leaf yield is assumed to be 55.4 % of the annual aboveground shoot yield for a given year. Daily leaf growth is assumed to occur during the defined growing period (May 1 – October 15) so annual leaf production is averaged across the growing period to provide data for daily intervals.

A similar approach was used to determine the amount of root biomass produced, where a fixed ratio of root to shoot production was used based on field data also obtained in 2009. Based on soil core data measured in 2009 (Cardinael et al. 2012) the total belowground biomass was estimated to be 1300 kg ha^{-1} yr^{-1}, which results in a root:shoot ratio of 0.43. Although Cannell (1988) observed that the proportion of biomass allocation from production varied for leaves (10-50%) and roots (5-30%), when the proportion of biomass allocation is averaged for the entire season the leaf:shoot ratio (0.62) and the root:shoot ratio (0.46) are comparable to the total ratio observed at the end of the growing season in Guelph. This suggests that the ratios are adequate for estimating total annual production of leaf and root biomass. The proportion of root biomass growth allocated to fine roots (67%) each year was assumed to remain constant over the 21-year lifespan and was based on 2-year field trials measured by Stace (1988) where an average fine root to coarse root ratio (FR:CR) of 2.025 was observed. Daily growth of fine root and coarse root biomass is derived using the same method as leaf biomass, where the annual root production for each fraction is averaged across the growing period at daily intervals.

4.2.2.4.4 Biomass Turnover

Leaf turnover is often reported in units of yr^{-1}; however, the turnover rate can vary significantly depending on the time of year. While leaf turnover is ongoing during the growing season, it increases significantly during the final few months of growth. Litterfall data from Guelph was measured during two distinct time periods: a long early growing period (June 11 – Sept 11) and a shorter leaf drop period (Sept 12 – Nov 20). The total average litterfall measured during the early period (251 kg ha^{-1} yr^{-1}) was approximately 15% of the total litterfall measured (1673 kg ha^{-1} yr^{-1}), while the leaf drop period accounted for approximately 85% of the total litterfall (1422 kg ha^{-1} yr^{-1}) (Cardinael et al. 2012). As a result, 15% of the annual leaf biomass produced is assumed to drop between June 11 and September 11, while the remaining 85% of leaf biomass produced is assumed to drop between September 12 and November 20. The total mass of litterfall released during each period is assumed to be consistent within each period and so total litterfall is averaged over the total number of days in the period to allocate leaf turnover at daily intervals.

Root turnover of fine root biomass is significantly higher than coarse roots; the former are generally regarded as having the highest turnover rates of all willow components (Rytter 1999; Stadnyk 2010). Coarse roots and biomass associated with willow stools are generally more stable over time (Pacaldo et al. 2013), and some fluctuations in coarse root mass can also be attributed to seasonal non-structural carbohydrates as per Rytter (2002), who observed up to a 60% increase in coarse root biomass during these months. For this LCA, it is assumed that there is little to no net turnover of coarse roots and stools, and so turnover of this biomass compartment is assumed to be zero. Coarse root biomass are assumed to increase over time until maximum yields are attained in the model, and will then remain stable at a fixed ratio relative to maximum aboveground yields (0.33) for the remainder of the life cycle.

The fraction of fine root biomass that turns over every year was assumed to be 0.85 by Grogan & Matthews (2002) in their model of soil carbon changes under SRC willow with frequent coppicing management. More recent work by Stadnyk (2010) of root biomass dynamics in willow trials in Saskatchewan found similar high turnover rates ranging from 0.9-1.1 yr^{-1}. For this LCA, a baseline fine root turnover rate of 0.9 yr^{-1} is used, and a sensitivity analysis of 0.8 – 1.0 yr^{-1} is evaluated. To simplify calculations of fine root turnover, the fraction of fine roots lost to senescence is assumed to happen the day after the end of the growing season.

4.2.2.4.5 Biomass Residue Characteristics

In order to sort and determine the amount of residues contributed to surface and soil carbon pools, inputs are required that indicate the timing and mass of litterfall and roots that enter those pools, as well as their carbon and lignin content in order to operate the soil carbon module. Foliar carbon content was measured from litterfall collected at the Guelph site in 2009 and the remaining carbon and lignin content of willow litterfall and woody material (stems and roots) were obtained from previous studies. The values used for inputs to the residues module and the sources from which they were obtained are listed in Table 4.5.

Table 4.6. Carbon and lignin content of willow leaves, stems, and roots. All sources that values are obtained from are listed for reference. Where a range was produced, the average value used is shown in brackets.

Willow Component	Lignin (%)	Source	Carbon (%)	Source
Leaves	22.1-27.6 (24.9)	Slapokas & Granhall 1991	47.9-51.9 (49.9)	Guelph field data
Wood	13.2	Szczukowski et al. 2002	50.8	Stolarski 2008
Fine Roots	25.4-35.1 (35.1)	Puttsepp et al. 2007	46.6	Stolarski 2008

4.2.2.5 Soil Carbon Module

Direct land use change effects on soil carbon balance and CO_2 emissions from soil are calculated using one of three different approaches. First soil carbon changes are calculated based on IPCC Tier I guidelines; a conservative approach is taken by assuming that there is no net change to soil carbon concentrations. Short-rotation coppice is often assumed to sequester carbon in soil at a faster rate than decomposition, and is considered a conservative approach where little to no data is available (see Buratti et al. 2013; Gonzalez-Garcia et al. 2013; Budsberg et al. 2012).

Net changes to soil carbon can also be calculated using a pre-defined emission factor that has been used in previous LCAs of willow feedstocks. Short-rotation willow is often found to sequester more carbon in soils than they emit over the lifetime of the plantation; Agostini et al. (2015) found an average sequestration rate of 0.63 – 0.72 Mg C/ha/yr across literature. For the second calculation approach, the baseline annual sequestration rate of carbon used in this LCA is a conservative 0.5 Mg C/ha/yr, and a range of 0.3 – 0.7 MgC/ha/yr is used for a sensitivity analysis.

The third method employed to evaluate soil carbon changes is through the use of a daily time-step modified version of the soil carbon model CENTURY (Parton et al. 1993). An established daily-modified version (DAYCENT) has been used in a few bioenergy LCAs previously (see Kim et al. 2009; Kim and Dale 2008; Adler et al. 2007), suggesting that the model can be employed successfully using finer-scale temporal data. A critique of the DAYCENT version is the quantity of data needed to both initialize and run the model, where complex crop growth models are combined with a soil carbon, nitrogen, and phosphorous model. The data requirement limits the application of DAYCENT where data are missing or limited (Lockwell et al. 2012). For the modified daily time-step version of CENTURY used in this LCA, only the soil carbon and residue decomposition submodels are used. The default biomass production submodel and soil moisture and temperature components are disconnected from the existing CENTURY model and have been replaced with modified versions discussed elsewhere in these methods.

While there are other soil carbon models that can be used at daily, weekly, and monthly time-scales, the existing proven abilities of the CENTURY model to be linked with other submodels was another deciding factor in its use. For example, while CENTURY includes a nitrogen submodel that can include inputs from residues, other models (RothC, APSIM, ICBM) don't have a nitrogen component integrated with them (Goglio et al 2015; Ranatunga et al 2001). Additionally, underestimation of CO_2 emissions from soil respiration were observed by Herbst et al (2008) using RothC, where biomass production and residues were not connected to soil dynamics.

The overall soil carbon flow is shown in Figure 4.3 and follows the same general structure as the original CENTURY model. Consistent with the SOM sub-model described in previous

CENTURY literature (Parton et al. 1988; 1993), all aboveground and belowground biomass residues inputs to soils are partitioned into metabolic or structural pools, based on a linear function of the lignin:N ratio of each residue pool. As the lignin:N ratio increases, more of the residue is partitioned to the structural pools which have relatively slower decay rates; the equations used to partition residues can be found in Appendix 4.3.

The decomposition of biomass residue and subsequent transition into soil carbon pools depends on whether the biomass is in the structural or metabolic pools. For carbon leaving the soil active pool, there are three different potential flows, including respiration by microbes, movement into the slow SOM pool and movement into the passive SOM pool. As in the CENTURY model, the decomposition rates are associated with carbon flow using general first-order decay equations as listed in Parton et al. (1993). For the decomposition of all carbon pools except the structural compartments and the active soil compartment, the rate is dependent on a fixed decomposition rate (K), and an abiotic factor (A) that is dependent on soil moisture and temperature (see Appendix 4.4). As in previous first order decay models, it is assumed that all C decomposition flows are associated with microbial activity and resulting respiration occurring simultaneously which is represented by flows illustrated in Figure 4.3.

All equations used to calculate decomposition rates and transfer between pools are described in detail in Appendix 4.4 and follow those described in Parton et al. (1988; 1993). The fraction of carbon lost as CO_2 during the transition between pools are listed in Table 4.6 along with the decomposition rates (K values). Default decomposition rates were adjusted from the default monthly values to reflect daily rates similar to modifications made by Grogan and Matthews (2002). Decomposition rates were also adjusted to reflect changes to the soil moisture and soil temperature factors which can either promote or inhibit decomposition rates. The effect of soil temperature on decomposition is described by Parton et al. (1987) and modified by Kirschbaum and Paul (2002).

The decomposition rate dependent on temperature is described by:

$$M_T = e^{\frac{3.36*(T-40)}{(T+31.79)}} \tag{2}$$

where T is the soil temperature. Several methods are available to measure and estimate soil temperatures. For this model, a simple linear method of relating air temperature to soil temperature, as described in methods by Parton (1984), is used, where the mean soil temperature (T_{SM}) each day is described as:

$$T_{SM} = 0.78 * T_{SM7} + 2.87 \tag{3}$$

where T_{SM7} is the 7-day mean air temperature. The effect of soil moisture on decomposition is based on the equation described in the daily version of the CENTURY model (DAYCENT) which is given by the equations:

$$M_M = \frac{1.0}{1.0+30*e^{(-8.5*RAT)}} \tag{4}$$

if RAT > 1.5, then

$$M_M = 1.0 - 0.7 * \frac{(RAT-1.5)}{1.5} \tag{5}$$

where RAT is the ratio of the soil water balance to the potential evapotranspiration for each day.

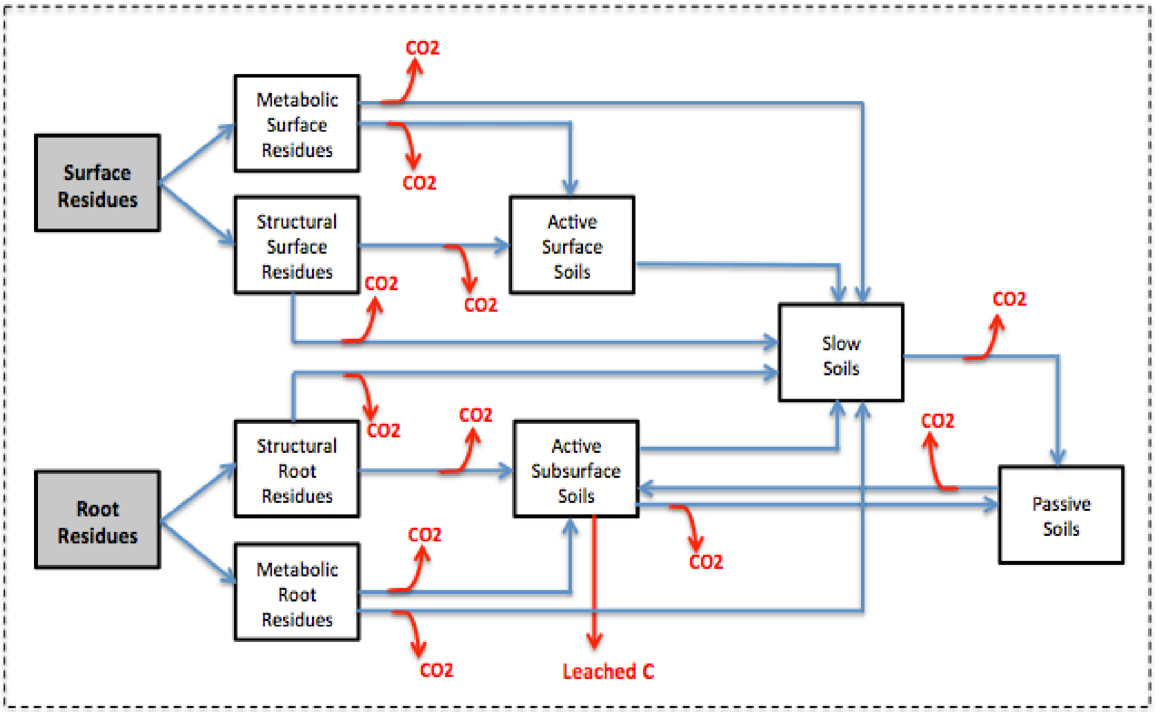

Figure 4.3. Soil Carbon Model flows and decomposition. Blue arrows indicate carbon flows between pools and are associated with decomposition represented by red arrows showing the release of carbon dioxide. Surface and root residues are partitioned based on fractions of lignin which separate into metabolic and structural residue pools.

Table 4.7. Decomposition fractions and k values for soil carbon and residue decomposition rates employed in the soil carbon sub-model.

Soil fraction	K_i	**K value**	F_{CO2}
Surface Active Pool	K_4	0.03	0.60
Soil Active Pool	K_3	0.04	0.53
Soil Slow Pool	K_7	0.001	0.55
Soil Passive Pool	K_8	0.000038	0.55
Metabolic Surface Pool	K_5	0.08	0.60
Structural Surface Pool for Active	K_1	0.02	0.60
Structural Surface Pool for Slow	K_1	0.02	0.3
Metabolic Soil	K_6	0.1	0.55
Structural Soil for Active	K_2	0.027	0.55
Structural Soil for Slow	K_2	0.027	0.3

4.2.2.5.1 Soil Moisture and Temperature Sub-model

Soil moisture and temperature have important controls on the decomposition of carbon in the soil carbon module; in the Century model, equations used to calculate decomposition requires an estimation of both parameters and so the Soil Moisture and Temperature Sub-model is designed to make estimations to make those calculations possible. Soil moisture also has a feedback effect upon itself through the rate of evapotranspiration (ET_o) that can decrease as soil moisture decreases and becomes less available to plants. The soil water balance (SW_i) for the LCA model used is described as:

$$SW_i = SW_{i-1} - ET_C - LW + P \tag{6}$$

where SW_{i-1} is the soil water balance (mm) of the previous time period, ET_C is the crop specific evapotranspiration (mm/day), *LW* is the amount of water leached as soil water content exceeds field capacity (mm/day), and *P* is the precipitation (mm/day).

For precipitation inputs (P), daily historical weather data for years 2006 – 2014 were obtained from the University of Waterloo Weather Station Data Archives. To simulate variation in weather for years 2015 – 2026, annual precipitation data selected at random from the 2006 – 2014 period were used in place of future data.

The amount of water that is leached (LW) from the soil incorporates any volume of water above soil field capacity. An assumption made in the model calculations is that when daily temperatures are less than or equal to 0°C, leaching is disabled. Since leaching is disabled when temperatures are below freezing and soils are assumed to be at field capacity at the beginning of each growing season, precipitation in the form of snow was not included in the precipitation datasets. Soil texture, bulk density, and field capacity were all measured on-site at the Guelph Turfgrass institute and are listed in Table 4.8. The plant wilting point (PWP) and readily available plant water (RAPW) are also listed in Table 8 and are based on calculations described in Allen et al. (2005). A full description of equations used to calculate plant-water values can be found in Appendix 4.5.

Table 4.8. Soil texture and water holding capacity of soils measured at Guelph Turfgrass Institute, May 2006.

Parameter	Value
Sand (%)	0.534
Clay (%)	0.33
Silt (%)	0.136
Bulk Density (g/cm^3)	1.25
Field Capacity (mm)	450
Plant Wilting Point (mm)	180
Readily Available Plant Water (mm)	90.45

To obtain the crop specific evapotranspiration rate (ET_C) for the soil balance equation, the actual evapotranspiration rate is expressed as a product of potential evapotranspiration, a crop-specific coefficient (K_c), and a linear soil water scalar index factor:

$$ET_C = ET_0 * K_c * \mathrm{Ks} \tag{7}$$

Potential evapotranspiration rates (ET_o) are largely controlled by air temperatures, incoming solar radiation levels, humidity, air speed, and the transpiration rate of specific crops. While the Penmen-Monteith equation is widely used to calculate evapotranspiration, it requires significant input parameters that are not always available or accessible for all locations such as air speed and humidity. Temperature and precipitation data are widely available from weather stations so evapotranspiration was calculated using an equation proposed by Hargreaves and Samani (1982, 1985) that allows evapotranspiration to be calculated using only daily minimum, maximum, and average temperatures combined with radiation input based on latitude:

$$ET_0 = 0.0135 * R_s * (TC + 17.8) \tag{8}$$

where R_s is the solar radiation and *TC* is the average daily temperature (°C). While relative humidity is not explicitly represented in the equation, it is implicitly contained in the equation by the difference in the maximum and minimum temperatures used to calculate solar radiation (R_s):

$$R_s = KT * R_a * TD^{0.5} \tag{9}$$

where *TD* is the maximum daily temperature minus minimum daily temperature (T_{max}-T_{min}, °C) and linearly related to relative humidity (Hargreaves and Samani, 1982). The other variable *KT* is an empirical coefficient, and R_a is the extraterrestrial radiation (mm/day). A detailed description of equations used to calculate R_a and KT is described in Appendix 4.6.

Although Kc values are expected to be consistent across similar species, they can be widely variable between sites and even for the same species at different points in the growing season. Crop-specific coefficients of willow field trials calculated by Mirck and Volk (2009) showed fluctuations in values throughout the growing season that did not exhibit normal distribution behaviour as expected in crop evapotranspiration models discussed in literature (Allen 1995; Text 2008). Variations in observed Kc values were found to be significantly coupled to changes in the atmosphere, but also showed correlations in their response to soil moisture and even stem diameter, suggesting that at the canopy-atmosphere interface, willow and agroforestry fields might be behaving more as forests rather than grasslands or croplands as the stand ages (Mirck & Volk 2009). Due to the complexity of modeling dynamic Kc values for willow stands and the lack of data (to date) that might describe seasonal Kc variations, the average Kc value (1.20) measured in Mirck & Volk (2009) across all willow varieties for all sampling periods for the growing season (Apr 30 – Nov 15) is used for this equation.

As the soil water depletes below a threshold that varies between crop and plant species, evapotranspiration will be reduced as transpiration and available soil water decreases; in order to account for this change, a linear soil water scalar index (K_s) is applied to the ET_c calculation. When the soil water balance is within the available plant water range, there is no adjustment to ET_c, and K_s is 1. When the soil water balance falls below the plant threshold, K_s falls below 1, until it reaches the plant wilting point of the soil where it has a value of 0. The greater capacity of a soil to retain water allows the crop to have more opportunity to obtain water from the soil and can reduce the rate at which K_s affects evapotranspiration.

4.2.2.6 Soil Nitrogen Emissions

The Tier I IPCC Guidelines suggest applying an emission factor of 1.25% for all nitrogen fertilizers and residues applied to soils (IPCC 2006). For Scenarios 1 and 2, the IPCC guidelines are applied using this recommended emission factor. For Scenarios 3 and 4, emissions from nitrification and denitrification of nitrogen in soil are calculated using a Tier II method based on a recent approach developed for Ontario conditions (Rochette et al. 2008). The method takes into consideration regional soil types and textures as well as differences between annual precipitation and adjusted crop evapotranspiration rates to take into account soil moisture variations based on average Ontario conditions. Based on Ontario data, the emission factor derived from Rochette et al. (2008) was 1.7% of total N added to soils - which includes all residue inputs as well as synthetic N fertilizer inputs. The percent nitrogen by mass for all residues and synthetic N fertilizers used is listed in Table 4.8. The nitrogen content of willow leaves can vary between sites and willow clone types. In field measurements performed at Guelph, nitrogen in leaves ranged from 2.35 – 3.23% by mass, while in other studies it has ranged from 3 – 3.33% (Merilo et al. 2006; Rellis et al. 2004; Garten et al. 2011). As a result, an average of the observed range was used (3%) for this study.

Table 4.9. Nitrogen content of willow leaves, stems, and roots. All sources that values are obtained from are listed for reference. Where a range was produced, the value used is in brackets.

Willow Component	Nitrogen (%)	Source
Urea	46	Nagy 2000
Leaves	3	Average from Literature
Fine Roots	1.66	Puttsepp et al. 2007

4.2.3 Soil Carbon Model Validation

Unlike quantitative empirical studies focusing exclusively on soil carbon dynamics, where extensive long-term field samples are compared to modeled or predicted changes, the lack of long-term data available for LCA scenarios has led many bioenergy LCAs to use sensitivity or uncertainty analysis exclusively as an approach to evaluate modeled soil carbon changes (e.g. Sanscartier et al. 2014; Daystar et al. 2015; Sinistore et al. 2015). For this LCA, sensitivity analysis of inputs and parameters to the soil carbon model are demonstrated in the sensitivity analysis section later in the results; however, an attempt at comparing soil carbon changes were compared to previous similar observations and are discussed along with the results.

4.3 Life Cycle Impact Assessment Results

The results of the baseline steady-state LCA scenario show that over a 21-year period the total GHG emissions for SRC willow cultivated in Guelph are 50.1 t $CO_{2}eq$ ha^{-1} (2.39 t $CO_{2}eq$ ha^{-1} yr^{-1}), while the total energy use is 433.6 GJ ha^{-1} (20.6 GJ ha^{-1} yr^{-1}). Assuming an average annual harvestable yield of 7 odt ha^{-1} yr^{-1} of willow biomass results in a total harvest over the lifespan of the plantation of 147 odt ha^{-1}, which corresponds to a harvested energy value of 2910 GJ ha^{-1}

(139 GJ ha^{-1} yr^{-1}). Assuming that the carbon content of willow wood is 50.8%, the total carbon sequestered into harvested biomass is approximately 74.7 t C ha^{-1} over the lifetime of the plantation (3.56 t C ha^{-1} yr^{-1}). Based on the difference between the overall emissions and energy use and the amount of carbon and energy sequestered in biomass feedstock, the net GHG emissions and energy use for the baseline scenario resulted in a net carbon sink of approximately 24.5 t C ha^{-1} (1.17 t C ha^{-1} yr-^{1}) over the entire lifetime of the plantation.

Adjusting the calculation approach for the soil carbon, nitrogen, and biomass growth modules resulted in differences in both the quantity of overall emissions as well as the total carbon balance. First, applying the soil nitrogen emission factor to crop residues as well as synthetic N applications increased total emissions from soils by 22.7 t C ha^{-1} over the lifetime of the plantation. Adjusting soil carbon calculations to include dynamic daily estimations of soil carbon balances did not change the overall energy use or total emissions, but did increase the amount of carbon sequestered as it included soil carbon in addition to biomass carbon. Total net organic carbon change over the 21-year period was estimated to be 8.71% in soil organic carbon. In all variations of the LCA model, the amount of carbon sequestered in biomass and soils is more than the carbon released through all emissions combined.

4.4 Life Cycle Interpretation

4.4.1 Contribution Analysis

Emissions from the transport of nursery cuttings are the primary source of GHGs in the steady-state scenario, accounting for approximately 55% of GHG emissions and up to 83% of total energy used from the upstream production stage, upstream transport stage, and the land management module of the biomass production stage (Figure 8). The large proportion of emissions is associated with the poor fuel efficiency of the transport truck (40L/100km) and the distance traveled each way (125 km); improving either of these variables can significantly reduce the contribution of the transport portion to total impacts. For example, reducing the distance to transport willows by half each way (62.5 km) would reduce total GHGs by 13.7 t, while improving the fuel efficiency of the transportation used by 20% would further reduce emissions by 8.2 t of CO_2*eq*.

Reducing transport distance by half would shift attention to emissions from soil nitrogen, which would become the major emission source within the system (44%); willow transport contributions would drop to only 38% of total GHG emissions. Decreasing transport distance and improving fuel efficiency by switching to full-ton pick-up trucks for transport simultaneously would reduce the impact contribution of transport even further; in this scenario transport accounts for only 23% of total GHG emissions and just over half (54%) of total energy use. The distance between field and upstream products is clearly critical to the overall impacts and emphasizes the importance of choosing sites in proximity to supply chains and choosing appropriately scaled transportation choices based on the size and scope of willow plantations.

When the contributions of transportation are removed from the comparison, the on-farm impacts are primarily associated with emissions from soil nitrogen (28%) and emissions from upstream nursery operations (37%) (Figure 4.4). Incorporating crop residues into the soil nitrogen module increased total emissions contributions so that 81% of all on-farm emissions are associated with soil N emissions, while nursery operations are reduced to 10% of total emissions (Figure 4.5). Not including residues as part of the emission factors has the potential to underestimate emissions by nearly 16 t CO_2*eq* ha^{-1} over the lifetime of the plantation. The higher emission factor used in the Ontario-specific version of the nitrogen module likely contributes to some of the higher emission values, but is justified given its calibration to Ontario climate conditions compared to global averages (Rochette et al 2008) and has been used for similar LCA calculations elsewhere (Kumudinie et al 2012). The remaining increase in emissions though is due to its application to the decomposition of residues in soils as Tier I emission factors only apply to fertilizer applications.

Applying emission factors to residues is appropriate and important to include because while it is true that some of the nitrogen released from residues is taken up by crops, some of it will also be released as soil emissions. Of particular interest and observed during this LCA was that most residue inputs to soil occur in the autumn during the leaf senescence period when biomass growth is beginning to slow down and nitrogen-uptake and use by crops are minimal; this has important implications for understanding the temporal patterns of how residues both contribute to emissions and supply nitrogen to soils and plants. For example, a number of studies (Burton et al, 2008) have found spring fertilizer additions to have significantly lower N_2O emissions when compared with autumn applications, likely a result of the ability of developing crop plants to utilize nitrogen, avoiding a shift in soil microbial populations that utilize nitrogen as an energy source; it is reasonable to expect that inputs from residues would have comparable temporal trends. The increasingly warm autumn periods due to climate change might then have the potential to release more nitrogen from residues prior to it being utilized by the plant, and so the nature of this relationship and its implications for emissions and fertilizer application rates should be looked at further.

When the soil carbon balance was evaluated using the dynamic model, it resulted in a positive net soil organic carbon change of +8.7% over the lifetime of the plantation. Soil carbon estimations show a net decrease in soil carbon in the first 5 years, which does not recover to initial SOC levels until year 9 (Figure 4.6). Field observations of willow elsewhere have observed a net loss of SOC until year 10 (Chen et al. 2013) which is consistent with the soil carbon levels estimated using the soil carbon module. Looking at the breakdown of soil carbon fractions modelled, it can be seen that there was not much overall change in the active or passive fractions, but that there was a gradual build-up of slow carbon fractions, likely a result of the input of lignin from above- and below-ground biomass residues (Figure 4.7).

While there are no long-term soil samples available from the Guelph site or anywhere else in Ontario to compare modeled soil carbon results, the trends in soil carbon change in these results were similar to those made from field observations elsewhere. For example, in a meta-analysis by Qin et al. (2016) of 20 datasets from studies that explored changes in soil carbon on cropland that had been converted to SRC willow, general trends over three periods of time show similarities to those modeled in this LCA (Table 4.9). Although values are not identical, the general pattern in soil carbon changes were consistent where there are initial drops in soil carbon following establishment before maximum yields are attainted but recover after approximately year 10.

Table 4.10. Observed average changes in soil carbon (%) under SRC willow from literature (Qin et al. 2016) compared with predicted soil carbon changes from this LCA.

Years	Qin et al. (2016) (% change)	Predicted (% change)
0 – 5	-5.0	-1.5
5 – 10	+6.0	+0.4
10+	+14	+5.5

In another study by Ens et al. (2013), SRC willow and subsequent soil carbon changes were measured in the third year following establishment at three locations in Ontario (Pickering, Guelph, Sault Ste. Marie). In the study, the average change in soil carbon was found to be approximately -0.3% of the initial soil carbon concentration; similar to Qin et al (2016) observations there is a loss of soil carbon after establishment, however the value modeled here (-1.5%) seems to reflect the values observed in field studies in Ontario more closely. The time it takes for soil carbon following SRC willow establishment to show positive changes (net sequestration) observed by Lockwell et al (2012) in Boisbrand, Quebec, was also found to be the same. With comparable average annual temperatures (6.8ºC) and precipitation (999 mm) to the Guelph site, both their observations and those modeled here reached a net positive change in carbon in year nine.

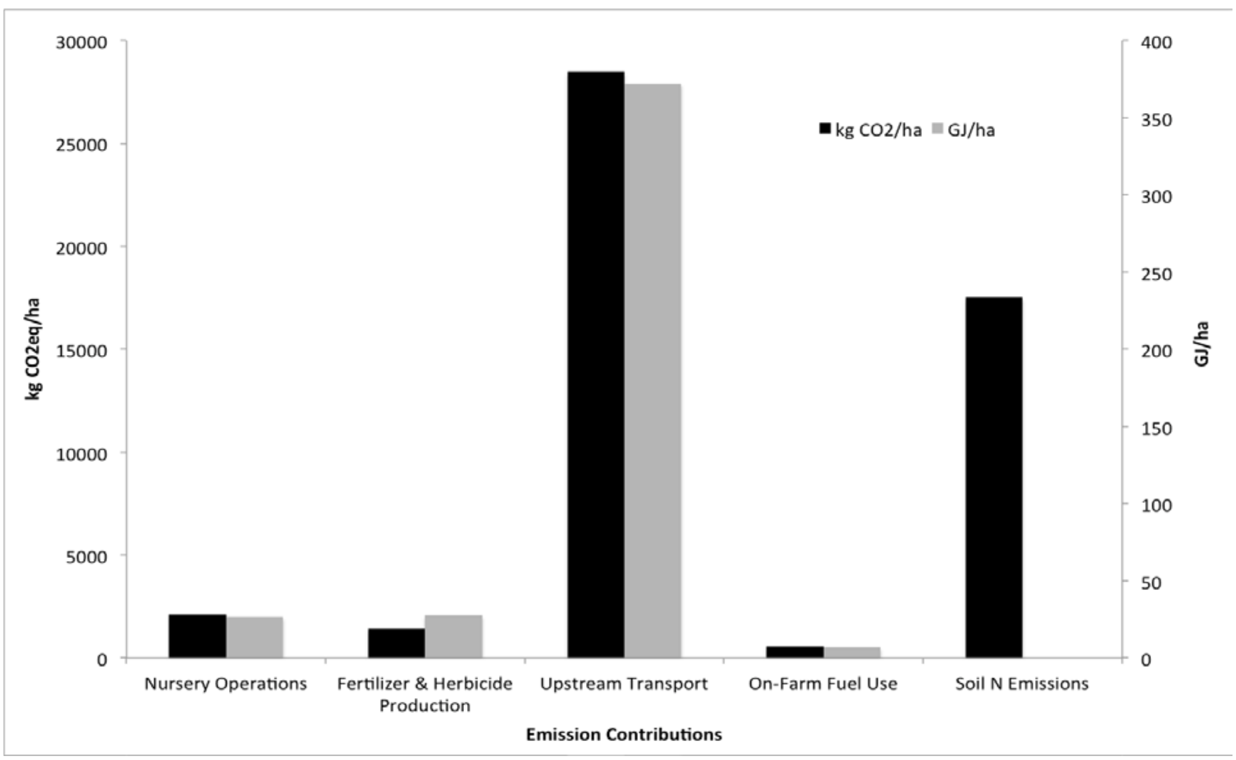

Figure 4.4. Total GHG emissions (kg CO_2/ha) and energy use (GJ/ha) from the upstream production stage, upstream transport stage, and the land management module of the biomass production stage.

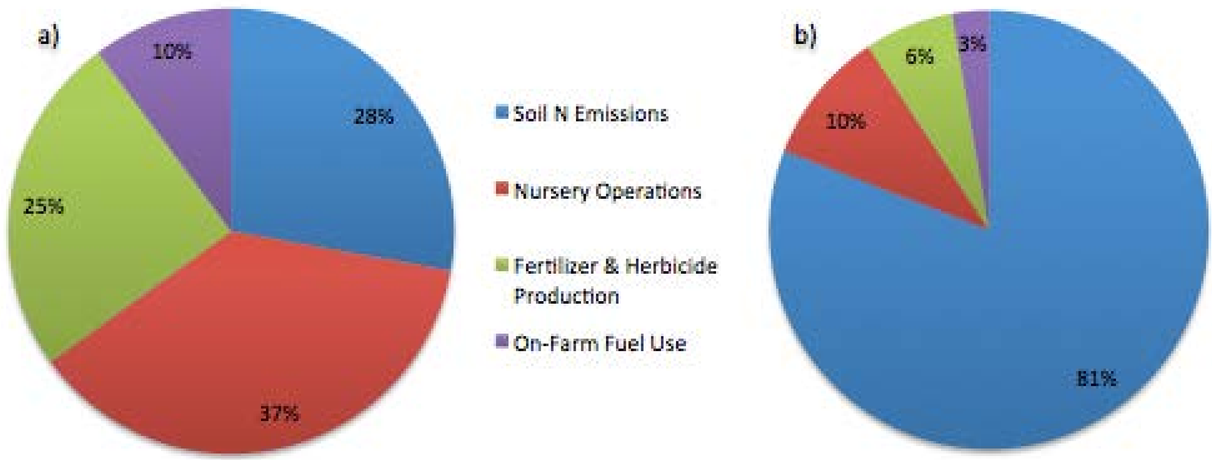

Figure 4.5. Contribution of processes to total GHG emissions (kg *CO2eq*/ha) over the lifetime of the plantation. Graph b) shows the difference in emissions contributions when soil N emissions include those from willow residues as compared to graph a).

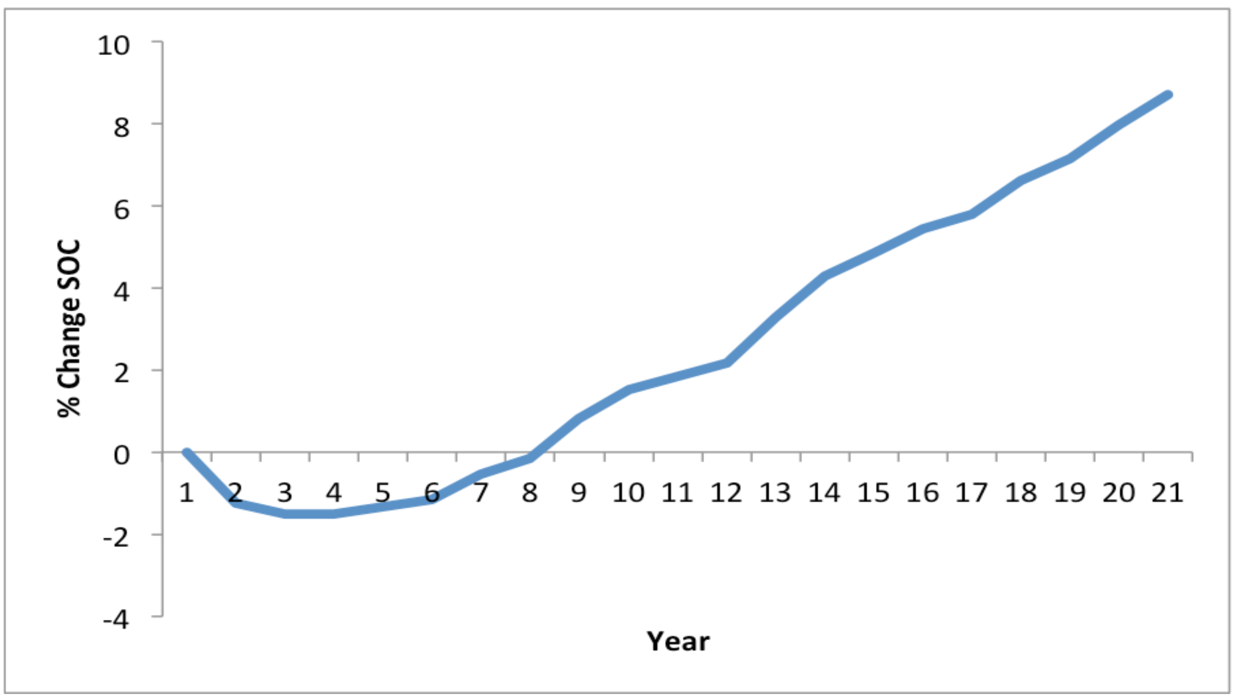

Figure 4.6. Line plot of cumulative % change in soil organic carbon relative to initial soil carbon concentrations.

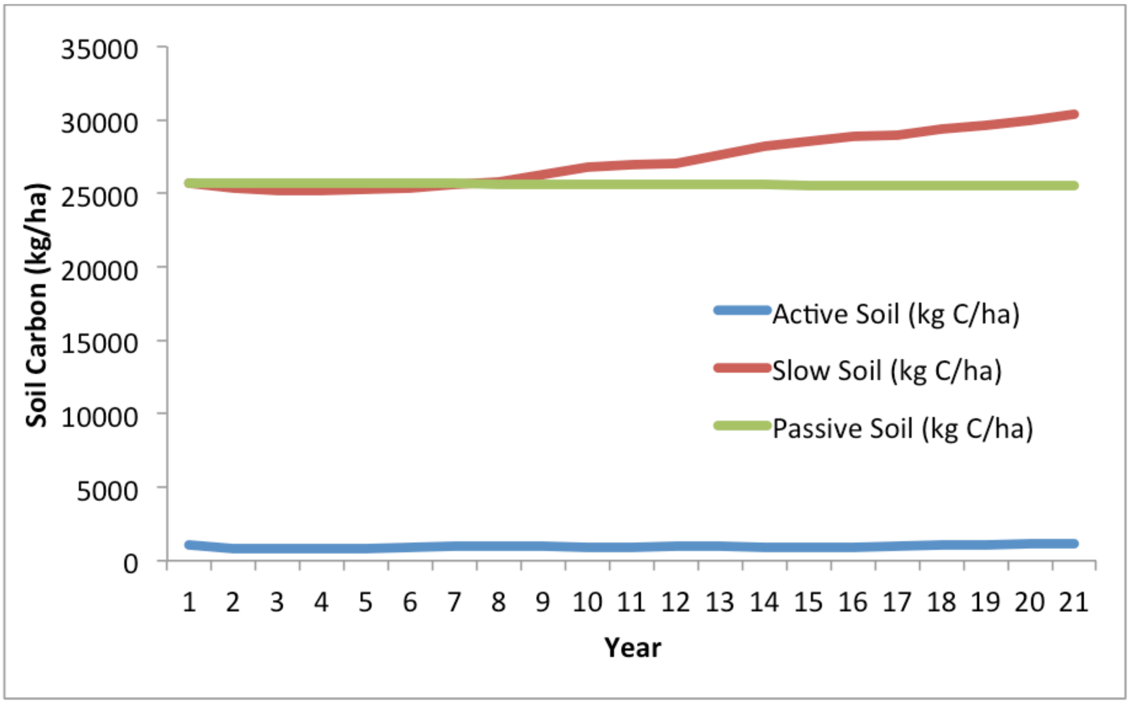

Figure 4.7. Line graph of overall soil carbon change along with changes in active, slow, and passive soil carbon pools over time.

4.4.2 Sensitivity Analysis

4.4.2.1 Average Willow Yields

The sensitivity of total harvestable yield figures ranges from 5 odt/ha/yr to 10 odt/ha/yr, around an average value of 7 odt/ha/yr. The effect of lower average yields (5 odt/ha/yr) is that soil carbon remains at a loss until year 14 and only shows a positive net change of +2.54% over the whole plantation lifetime. Not surprisingly, increased yields (10 odt/ha/yr) will decrease the time needed to recover soil carbon (net sequestration in year 6) and increase the cumulative % change in soil carbon to nearly +18%. Due to residues being the sole input of new organic matter to the soil system, alterations to the quantity of residues input to soils have significant effects on the net benefits or impacts to soil carbon.

4.4.2.2 Willow Characteristics

Given the importance of total residue inputs to soil carbon balances, the sensitivity of the model and of the soil system to changes in the values used to characterize willow residues was evaluated. The leaf:shoot ratio, root:shoot ratio, and the proportion of fine root mass that turns over were all varied by 20% and the changes to the final soil carbon balance (57074 kg C/ha) were compared (Table 4.10). Overall variations to the leaf residue portions seemed to have the largest influence on soil carbon changes, likely due to the higher volume of aboveground residues input to soils relative to belowground contributions.

Table 4.11. The quantity of carbon per hectare at the end of the 21-year life cycle are shown based on a 20% variation in the default value for each parameter. The percent change from initial soil carbon conditions are given in brackets.

Parameter	Default Value	-20% Variation ($kg\ C\ ha^{-1}$)	+20% Variation ($kg\ C\ ha^{-1}$)
Leaf:Shoot Ratio	0.554	55868 (6.415%)	58280 (11.01%)
Root:Shoot Ratio	0.43	56012 (6.69%)	58139 (10.74%)
FR Decay Fraction	0.85	56983 (8.53%)	57135 (8.82%)

To evaluate the sensitivity of soil N emissions from residues, variations in the assumed nitrogen content of residues (leaves and fine roots) were varied by 20% and compared to the total emissions from default values (15909 kg CO_2eq/ha). Similar to trends in the soil carbon balance observed with varying leaf fractions, varying leaf nitrogen by 20% resulted in the largest shift in total emissions from soil nitrogen, with a range of approximately 5000 kg *CO2eq*/ha over the lifetime of the plantation (Table 4.11). Again, the larger proportion of residues from leaves is likely the reason for the larger influence on total emissions.

Table 4.12. The quantity of nitrogen emitted per hectare at the end of the 21-year life cycle are shown based on a 20% variation in the default value for each parameter. The percent change from initial nitrogen conditions are given in brackets.

Parameter	Default Value	-20% Variation (kg C ha^{-1})	+20% Variation (kg C ha^{-1})
Leaf % N	3.0	13433	18384
Fine Root % N	1.66	15202	16615

4.4.2.3 Variable Willow Yields

Realistically, willow yields vary by year and are generally lower in the establishment years of plantations; the sensitivity of the soil carbon and nitrogen emissions to temporal variability of yields was assessed. Each variation was performed so that the total average yield over the 21-year lifetime remained equivalent to 7 odt/ha/yr. Following input of each of the yields, the total changes in soil carbon and soil N emissions were calculated (see Appendix 4.7).

A regression analysis of the final soil carbon content at the end of the 21-year life cycle compared with the average willow yield input showed that there was little correlation between the two parameters. The results suggest that annual variability between years can affect overall soil carbon dynamics even though average yield over the 21 years are nearly identical. For example, four of the years had an average annual yield of 7.05 odt/ha/yr but showed variations in soil carbon balances of more than 0.3 t C/ha; the difference in the time taken to recover soil carbon losses varied by almost a year depending upon performance. These results suggest that the sensitivity of the soil carbon model to yield changes can happen on annual time-periods and that averaging out yields can smooth over these dynamics.

4.4.3 Limitations and Data Gaps

The LCA results show that the largest contributor in the current analysis is the transportation distance and mode of transport for upstream products, as well as emissions resulting from the release of nitrogen in soils. Very little data was available to describe alternative supplier locations and transport options, and so different configurations were not tested. It may be assumed, however, that improved transport data and more ideal supplier locations might shift the burden of impacts to on-farm soil emissions. Given the sensitivity analysis of the soil N module to residue inputs, it is important to get accurate details of both the quantity of residues being input to soils as well as the nitrogen content of the residues. While leaf biomass and fine root biomass were assumed to be 3% and 1.66% N by mass respectively, these can vary depending on the type of willow clones grown (Chen et al. 2013) and on the characteristics of the soil being used (Liska et al. 2012; Mola-Yudego 2009). As a result, improved documentation of variations across willow clones in terms of nitrogen content and the conditions under which those might change are important variables to track in willow databases for future assessments.

Sensitivity analysis of the soil carbon model to changes in yields and the dynamics of those yields over time suggest that better coupling between realistic willow yields and soil carbon dynamics are important. A significant limitation for this study is the lack of long-term yield data and long-term soil carbon data; this limits the accuracy of the yields being estimated and limits the accuracy of the residue estimates being used as inputs to the soil carbon model. There is a distinct need in Ontario for long-term yield data, as well as data on the amount of residues input from those yields. This data could better explain the response of soil carbon to residue inputs and the ratio between soil carbon and yields. While this LCA showed that net changes to soil carbon are likely to be positive, how long it takes for sequestration to begin and what role variations between years play in the rates of soil carbon change are not certain. Consequently, over-estimation of yields in the formative years of the plantation might lead to over-estimation of residue contributions, delivering results that might suggest carbon sequestration is achieved at a rate faster than is actually occurring.

Sensitivity analysis of the various input parameters to the soil carbon model also showed that while soil carbon changes seem to remain positive, there can be significant differences over time, and as a result there can be significant differences in overall results, especially when small temporal variations in yields are compared. Willow yields in Kopp et al. (2001) followed a general logistic curve function, but there was significant variation in the annual yield which seemed to be susceptible to changes in precipitation and temperatures, particularly in the years following coppicing (Kopp et al. 2001; Liu et al. 2013). Consistent annual yields may give the impression that annual supplies are also consistent; actual increases or decreases in annual yields can affect annual residue contributions and have downstream implications in terms of energy supply and demand. It is important to model these interactions more accurately. While these results don't seem to change the environmental sustainability of deploying willow, they do suggest that there is a wide range in how sustainable these systems are. These may be critical differences that need to be accounted for if we are comparing differences in deploying alternative land-use options.

Another important implication of the sensitivity of soil dynamics to yields and residues are potential long-term feedback effects. Without long-term data, it is difficult to anticipate soil carbon sequestration patterns over time. It is also important to have data describing very fine temporal scales, particularly in fields that can have heterogeneous soil patterns where multiple management approaches may need to be employed for the same field (Santori et al. 2010). These imply that maintaining more spatially and temporally explicit databases across a range of geographical regions could inform more sustainable site-specific practices, especially in systems exploring stover and other residues as potential bioenergy feedstocks.

4.5 Future considerations for assessments of biomass systems

Previous assessments of bioenergy systems have described the need for the use of fertilizers produced from more sustainable pathways such as production of ammonia from bioenergy (Ahlgren et al. 2008). Based on estimates from this LCA, the dynamics of perennial systems - where fertilizers are applied more sparingly than with annual agricultural crops - suggest that emissions from residue inputs are likely to contribute more to cumulative impact than synthetic fertilizer applications over the entire life cycle. Choosing which perennial crops (or in the case of willow, which clone variety) should therefore take into consideration the nitrogen uptake by each variety, and the potential for subsequent re-input to the soil system through residue contributions. Selecting varieties that have lower nitrogen needs and concentrations, or selecting those with higher lignin ratios, could reduce the amount of GHGs emitted from decomposing residues at the farm-scale.

When inventory data is collected to fit the functional unit rather than more appropriate fine scales of unit processes, it can become difficult to represent changes in those processes or underlying modules within specific life cycle stages or groups of processes. Since it has been identified that LCAs need to be able to incorporate improvements to existing or new models as they are developed in order to allow for both bottom up and top down input, the modular approach shows that incorporating processes at varying time scales can improve LCA results. For example, the soil carbon module was calculated using three different approaches to temporal variability, which not only could improve estimations of soil carbon emissions but also expanded the ability of the LCA to evaluate a new environmental indicator, soil carbon balance (a function of soil quality). Future modulation of additional areas of the bioenergy system could also be applied such as improved estimations of eutrophication effects whereby data inputs that denote the timing of fertilizer events could be used to improve estimations. This shows that creating a dynamic database that can keep track of all spatial and temporal data, whether it is currently being used or not, has value for long-term evolution of LCA estimations.

There are some emerging dynamic models that have attempted to estimate willow growth at fine temporal scales (e.g. days or hours) in the context of changing environmental conditions. For example, Wang et al. (2015) use a leaf carbon exchange model to describe net production and links biological responses to environmental changes in temperature, light, and moisture; their work borrows from the leaf phenology work done by Arora & Boer (2004) where leaf development and mass allocation is coupled to fine-scale changes in environmental conditions. These types of models offer the potential for robust estimations that can be integrated with a number of site-specific parameters. They are currently limited in their capacity to inform long-term LCA inventories due to the short temporal period (1 year) they have been verified for, but also in the availability of site-specific data and parameters needed to operate the models (a pre-existing issue with dynamic LCAs already). The other limitation in applying models designed solely for willow growth to agroforestry practices is that they have yet to be fully integrated with land management actions and environmental changes which can have complex relationships.

4.6 Conclusions & Recommendations

A dynamic model framework for LCA was developed that couples LCA methodology to soil dynamics, biomass growth, and land management activities via a modular geographical database to quantify potential environmental impacts of SRC willow biomass feedstocks. Results show that transportation and emissions from soil nitrogen processes are the largest contributors to the GHG emissions and energy use. Soil carbon balances were estimated using fine-scale, site-specific data; results suggested that a net positive change in soil carbon was likely for all sensitivities of the model. Sensitivity analysis revealed that the soil carbon model is extremely sensitive to residue inputs and residue characteristics. More data in the Ontario context are needed to verify the use of these models, but early results suggest that these types of dynamic models can prove useful in understanding contributions of changing system inputs to overall LCA results. Overall, the LCA GHG emissions were similar to those found in the literature but the modular approach helped identify three key findings:

i. **Upstream suppliers of system-specific inputs such as willow cuttings are not yet widespread; there location and transport method chosen for delivery have significant overall impacts on the willow system**

Yields and biomass characteristics from newly developed willow clones are expected to improve as breeding programs expand. Development of a comprehensive database to document expected yields and residue characteristics would improve future dynamic model estimations. Fostering databases which better describe the geographic location of potential upstream inputs to willow production, and using this data to improve integration between supply and on-farm demand, could allow individual projects to better improve on their respective impacts.

ii. The soil carbon model used needs more robust validation data, as the module is very sensitive to changes in the system

While changes are not expected in the overall net soil carbon change, the rate and degree to which soil carbon is sequestered is hard to verify based on this LCA. This parameter needs to be evaluated using better regional field measurements.

iii. Results can better reflect the reality of small scale processes when the spatial and temporal attributes of small-scale processes are maintained

When the functional units and product system are represented at geographic scales that differ from the scale needed to understand unit process dynamics, the functional unit may fail to capture spatial and temporal sensitivity. This study showed that by structuring the LCA in such a way that processes can be calculated at multiple scales simultaneously, geographic attributes of the underlying data are not lost. Spatial and temporal variability of data and system dynamics can actually work with the functional unit if data is aggregated at the site, field, or sub-field level, rather than aggregating inventories to functional unit flows pre-analysis.

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4.8 Appendix

A.4.1 Upstream Fertilizer and Chemical Emission Factor Calculations

The upstream emissions and onsite mass balance additions associated with fertilizer applications are based on data listed in Nagy (2000), and are the 2006 values with the adjusted and improved emissions based on improvements to technology in the fertilizer production industry. Emission factors are used to evaluate GHG emissions based on the following equation:

$$E_{fert} = M_{FA} \times Ai \times EF_N \qquad [1]$$

where M_{FA} is the mass of fertilizer added (kg/ha), Ai is the amount of active ingredient per mass of fertilizer (a.i./kg), and EF_N is the emission factor for the amount of carbon equivalent emissions per mass of nutrient added (kg CO_2/kg N) or the amount of energy consumed per mass of nutrient added ($MJ_{consumed}$/kg N).

Table A.4.2 Variable Yield Estimates

Year	Predicted Yield (odt/ha/yr)	Simulated Annual Variation (% of Max Yield)	Simulated Variation of Predicted Yield (odt/ha/yr)
6	0.40	0	0.40
7	3.11	0	3.11
8	8.01	0	8.01
9	9.34	0	9.34
10	9.49	0	9.49
11	9.50	0	9.50
12	9.50	83	7.88
13	9.50	84	7.98
14	9.50	88	8.36
15	9.50	99	9.40
16	9.50	63	5.98
17	9.50	80	7.60
18	9.50	89	8.45
19	9.50	89	8.45
20	9.50	95	9.02
21	9.50	65	6.17
Average	8.68		7.38

A.4.3 Metabolic & Structural Biomass Carbon Fractions Calculations

Consistent with the original CENTURY SOM sub-model by Parton et al. (1987) and with early forest versions of CENTURY by Sanford et al. (1991), the aboveground and belowground biomass residues are partitioned into metabolic or structural pools, based on a linear function of the lignin:N ratio of each residue pool. As the lignin:N increases, more of the residue is partitioned to the structural pools, with relatively slower decay rates, as described by the original equations used by Parton et al. (1987), as Kirschbaum and Paul (2002) found no improvement or advantage to separating the biomass residue pools via any other permutation of the equations:

$$f_m = 0.85 - 0.018 * (\frac{N_{res}}{Ln_{res}}) \qquad (2)$$

$$f_s = 1 - f_m \qquad (3)$$

where f_m is the proportion of biomass residue classified as metabolic, f_s is the proportion of residue classified as structural, N_{res} is the concentration of nitrogen in residue, and Ln_{res} is the concentration of lignin in residue.

The biomass allocated to the structural pool is further divided based on the amount of lignin contained in the structural material, where 100% of all lignin in the structural biomass pool is allocated to the structural lignin flow, while the remainder is allocated to the structural cellulose and hemicellulose pools:

$$L_{Ln} = L_{struct} * Ln_{res} \qquad (4)$$

where L_{Ln} is the lignin fraction from the structural leaf residue pool, and L_{struct} is the total structural leaf residue. This equation is applied to all residues to create lignin fraction flows for structural pools from stems (S_{Ln}), fine roots (FR_{Ln}), and coarse roots (CR_{Ln})

A.4.4 Decomposition of Biomass Residue and Carbon in Soil

Following partitioning to specific residue pools, the decomposition of biomass residue depends on whether it is in the structural or metabolic pools. As in the CENTURY model, the decomposition rates are associated with carbon flow using general first-order decay equations, as described in Parton et al. (1993). For the decomposition of all carbon pools except the structural compartments and the active soil compartment, the rate is dependent on a fixed decomposition rate (*K*), and an abiotic factor (*A*) that is dependent on soil moisture and temperature:

$$\frac{dC}{dt} = K_i * A * C_i \tag{5}$$

where C_i is the carbon content of the residue or soil pool in question, A is the abiotic factor that is described later, and K_i which is the maximum decomposition rate (year^{-1}) for the residue pool in question. The decomposition of the structural pools follows a similar first-order decay, but also includes an additional factor:

$$\frac{dC_i}{dt} = K_i * L_C * A * C_i \tag{6}$$

where L_C is a factor that takes into consideration the fraction of structural C that is lignin, calculated as:

$$L_c = e^{(-3 * L_S)} \tag{7}$$

where L_S is the faction of structural C that is lignin in the originating state variable pool. The structural carbon pool follows two routes of possible decay as shown in Figure 1; either decomposition to the slow pool which includes all carbon from the structural pool that is lignin, or decomposition to the surface microbes that include mainly other celluloses and hemicellulosic material. For the soil active layer, an additional soil texture factor (T_m) is used instead of a lignin factor:

$$\frac{dC_i}{dt} = K_i * A * T_m * C_i \tag{8}$$

where T_m is calculated as:

$$T_m = 1 - 0.75 * T \tag{9}$$

where T is the silt plus clay content of the active soil fraction.

As in previous first order decay models, it is assumed that all C decomposition flows are associated with microbial activity and resulting respiration occurring simultaneously.

A.4.5 Partitioning of Carbon after Active and Slow Soil Decomposition

For carbon leaving the soil active pool, there are three different potential flows, including respiration by microbes, movement into the slow SOM pool and movement into the passive SOM pool, as described by Parton et al. (1987; 1993) and illustrated in Figure 1. The fraction of carbon that leaves as CO_2 (F_t) is given by the following equation, where T has already been defined:

$$F_t = 0.85 - (0.68 * T) \tag{10}$$

The fraction of carbon that is moved to the passive pool is described by:

$$C_{AP} = 0.003 + 0.032 * T_c \tag{11}$$

where T_C is the clay content of the soil active pool. The amount of carbon leached below 30 cm of soil is described by:

$$C_{AL} = \left(\frac{H_2O_{30}}{18}\right) * (0.01 + 0.04 * T_S) \tag{12}$$

where T_S is the sand content of the soil active pool, and H_2O_{30} is the volume of water that was leached beyond the 30cm depth. Finally, the remaining leftover carbon flow is moved to the slow fraction by the following equation:

$$C_{AS} = (1 - C_{AL} - C_{AP} - F_t) \quad (13)$$

For carbon leaving the slow pool, it can flow on to the passive pool (C_{SP}), or return to the active pool (C_{SA}), using the following equations from Parton et al. (1987; 1993):

$$C_{SP} = 0.003 + 0.009 * T_c \quad (14)$$

$$C_{SA} = (1 - C_{AP} - 0.55) \quad (15)$$

Passive SOM formation is a function of clay content, largely under the control of stabilizing active SOM, and to some extent the decomposition of slow SOM. The assumption that clay soils have a higher fraction of passive SOM is based on soil 14C data (Becker-Heidman 1989), and similar to the impacts of clay on SOM dynamics assumed by Jenkinson (1990).

A.4.6 Soil Water and Plant Water Calculations

To calculate the total water depth in *x* cm of soil, the volumetric water content of the soil (Θ_v) is calculated, using the bulk density (ρ_b) and mass water content (Θ_m) of the soil:

$$\theta v = \theta m * \rho_b \quad (16)$$

Using the volumetric water content of the soil and the depth of the soil (d_s) in question, the total water depth (d_w) currently in the soil is given by:

$$dw = \theta_v * d_s \quad (17)$$

The available water capacity (AWC) of a soil is the difference between field capacity (Θ_{FC}) and the permanent wilting point (Θ_{PWP}):

$$AWC = \theta_{FC} - \theta_{PWP} \tag{18}$$

To determine the total available plant water (*TAW*) for the plants roots, it is found by multiplying the AWC by the plant rooting depth (r_d):

$$TAW = (\theta_{FC} - \theta_{PWP}) * r_d \tag{19}$$

To determine the readily available plant water (RAW), the TAW is multiplied by a crop-specific allowable depletion factor (f_{AD}):

$$RAW = TAW * f_{AD} \tag{20}$$

Allen et al. (1998) described a way that f_{AD} values, which are estimated based on ET of 5mm/day, can be adjusted to reflect the actual ET:

$$f_{ADactual} = f_{AD} + 0.04 * (5 - ET) \tag{21}$$

To calculate the Ks value, first the readily available plant water (RAPW) is subtracted from the total available plant water (TAPW) in order to find the approximate volume of water available beyond the plant threshold. For example, if the TAPW is 36 mm and the RAPW is 24.12 mm, then there will be approximately 11.88 mm of water available beyond the defined plant threshold, but for which some decrease in availability will happen until the soil water availability reaches the plant wilting point. Since Ks has a value between 0 and 1, a negative linear function is calculated with the values 0 to 1 on one axis and the range of available water on the other (See Figure X).

A.4.7 Radiation & Temperature Calculations

Temperature and solar radiation are important parameters in estimating ET_0. Solar radiation (R_s) in this model is estimated based on the equation recommended by Hargreaves and Samani (1982):

$$R_s = KT * R_a * TD^{0.5} \tag{22}$$

where *KT* is an empirical coefficient, R_a is extraterrestrial radiation (mm/day), and *TD* is the maximum daily temperature minus minimum daily temperature (°C) for weekly or monthly periods.

The original equation to estimate solar radiation by Hargreaves and Samani (1985), can also be written as (Ugwa and Ugwuanyi, 2011):

$$R_s = KT * \sqrt{T_{\max} - T_{\min}} * R_a \tag{23}$$

where T_{max} and T_{min} are the daily maximum and minimum air temperatures (°C) respectively.

The extraterrestrial radiation, R_a, for each day of the year can be calculated based on the latitude, solar constant, solar declination and the time of the year, based on equations set out in Allen et al. (1998):

$$Ra = \frac{24*60}{\pi} * G_{SC} * d_r * [\omega_s * \sin(\varphi) * \sin(\delta) + \cos(\varphi) * \cos(\delta) * \sin(\omega_s)] \tag{24}$$

where

R_a = extraterrestrial radiation (MJ/m2/day)
G_{SC} = solar constant (0.0820 MJ/m2/min)
d_r = inverse relative distance Earth-Sun
ω_s = sunset hour angle (radians)

R_a, expressed as MJ/m2/day, can be converted to the equivalent evaporation in mm/day by multiplying Ra by 0.408. The conversion from decimal degrees to radians can be given by the following equation:

$$[Radians] = \frac{\pi}{180}[decimal \deg rees] \tag{25}$$

The inverse relative distance Earth-Sun, *dr*, is given by:

$$dr = 1 + 0.033\cos\left(\frac{2\pi}{365} * J\right) \tag{26}$$

and the solar declination, δ, is given by:

$$\delta = 0.409\sin\left(\frac{2\pi}{365} * J - 1.39\right) \tag{27}$$

where J is the number of the day of the year (1 to 365), and the sunset hour angle, ω_S, is given by:

$$\omega s = \arccos[\tan(j)\tan(\delta)] \tag{28}$$

where *j* is the latitude in radians, and the total number of daylight hours (N) are given by the equation:

$$N = \frac{24}{\pi}\omega_s \tag{29}$$

The empirical value *KT* has a recommended value of 0.162 for interior regions and 0.19 for coastal regions. Other factors besides solar radiation and temperature can influence the maximum and minimum temperatures for a given location, however, and for low latitudes the temperature difference can become negligible and could significantly underestimate both solar radiation and ET_0 (Jagtap, 1991). Using monthly temperature and radiation data for a period of 25 years in the continental United States between latitudes 7 and 50 degrees N, Knapp et al. (1980) found the following relationship between *TD* and *KT* across 65 weather stations:

$$KT = 0.00185 * (TD)^2 - 0.0433 * (TD) + 0.4023 \tag{30}$$

The relationship of the curve shows that in advective environments where the temperature difference may exceed 14^{o}C, there is a sudden rise in *KT* value, explaining the underestimation of ET_0 in dry climates when using equation 2 without *KT* adjusted.

Chapter 5 – Conclusion

5.1 Conclusion

This thesis is the first study of its kind to explicitly characterize how spatio-temporal attributes of data and system definitions are managed and applied in LCA and the potential implications it can have on results when it is used to evaluate supply chains that include environmental systems, such as bioenergy. Agroecosystems may perform differently as geography changes due to site-specific characteristics such as soil type and local climatic conditions, however the trends identified in the studies performed here highlight how LCAs of bioenergy crops are often performed using data collected at various spatial and temporal scales with little attention paid to fine-scale geographical attributes. The prominent use of data aggregated from large-scale datasets and the varying scales at which they were scaled from was shown to result in frequent discrepancies in results between LCAs and difficult to interpret in terms of informing small-scale land-management decisions. In attempting to bridge some of the gaps in understanding about how to include geographically-explicit approaches with LCA standards, this thesis was able to identify some of the ways in which this could be improved, but also provided direction on how vague or absent instructions in existing LCA standards might be altered to take into consideration these recommendations.

5.2 Identifying geographical concepts for integration into LCA

The first objective of this thesis was to identify specific geographical concepts that would be best integrated into life cycle assessment, based on key dynamics of bioenergy systems. A thorough review of existing LCA standards revealed that they do not provide direction in setting geographical parameters of system boundaries and unit processes, leaving those decisions up to the practitioner to interpret. It was determined that the concept of scale is absent from LCA standards and specifically data quality requirements where practitioners are advised to only address the geographical extent of product systems and their underlying processes. Understanding geographical phenomenon requires determining the resolution at which they occur, not simply the extent to which they may have an effect. System boundary scales need to be more explicitly defined during the goal and scope phase of the LCA and so there is a need for guidance on how they are set from a spatial and temporal attribute basis in order to ensure that critical information about finer-scale processes are not misrepresented.

It was demonstrated that most existing commercial inventory databases have limited types of feedstocks available; data on feedstocks that are available tend to be provided at resolutions ranging towards national or multi-national scales. There needs to be transparency about where the underlying data that makes up these databases originated; improved direction on setting geographical attributes of inventory and process data would be an important first step in advising LCA practitioners. If life cycle inventory data was organized so that spatial and temporal attributes of the data was clearly highlighted, it could facilitate an on-going iterative approach that could minimize loss of data as they are aggregated from farm-scale measurements to inventory averages and improve our understanding of impacts and land use change on more local and site-specific levels.

The ways that LCA impacts are aggregated and reported as midpoints and endpoints were also shown to conflate issues of geographic scales. LCA software that reports impacts in such ways can make it difficult to interpret complex systems, and so it may be more appropriate, in some instances, to address impacts for specific subsystems separately from the overall system in order to better reflect appropriate temporal and spatial scales of analysis. For example, the global warming potential midpoint provides little information about the impacts of processes at localized spatial scale or compressed temporal scale. Given the heterogeneous nature of biomass feedstock production across landscapes, this thesis highlights the need to address small-scale geographical attributes in LCA standards over the longer term in order to address many of the outstanding dynamic issues.

5.3. Critical assessment of scale as it pertains to LCA results & interpretations

The systematic review performed in this thesis exposed how a number of data quality issues can arise in LCA when assessments use data from a variety of scales; these can manifest as data is transferred to the inventory or when inventories are applied to impact assessment approaches. One specific observation made was that functional units are not explicitly described in all LCAs reviewed, which may be a result of vague directions in ISO 14044:2006 standards. There is a wide range of products and end uses possible using different bioenergy technologies and biomass feedstocks; LCA studies that focus on end-use products rather than intermediate products more closely linked to environmental processes (i.e. feedstock growth) may not see the need to address the more environmentally-focused ISO 14044:2006 standards. It was determined that because of the eventual need to compare different bioenergy streams or different energy pathways, there is a need to be explicit when defining the functional unit.

A second observation is that when functional units were explicitly stated, there was significant variation as to whether geographical attributes (spatial or temporal) were chosen to be included. There is an immediate need for more consistency in the way that geographical attributes are chosen to be applied to functional units. In the review, it was found that arbitrary inclusion of geographical attributes such as per hectare or per year made comparisons between other LCAs difficult and limited understanding as to what the information being presented actually means. It was advised that geographical attributes in the functional unit should only be used if it is important to reflect impacts relative to specific spatial or temporal scales, and that these relationships should be clearly and adequately justified. It seems apparent from the results of the review that practitioners are using geographical attributes in functional units as a means to relate the resolution of impacts to the extent of the product system. It may be appropriate to address this issue in LCA standards by providing better direction in defining the relationship between the functional unit and the product system in terms of geographical terms, particularly if they are referring to different scales of observation.

There was a similar disconnect observed in geographical definitions when it came to cut-off criteria used to define the spatial and temporal boundaries of system processes. These criteria were found to lead LCAs to overlook important impacts at fine spatial or temporal scales. The definitions for cut-off criteria are based on satisfying the geographical extent of system boundaries; the lack of discussion on satisfying resolution issues may be used to justify ignoring small-scale processes such as soil carbon dynamics. Inclusion of discussions of scale in cut-off criteria would make those decisions more transparent, without inhibiting many of the existing broad-reaching LCAs from being performed.

While it was previously understood that inventory data could be aggregated up or down from field or farm-scale measurements, it was found that the aggregated inventory data was often scaled more than once as impact assessments were calculated, particularly when geographical attributes were contained in the functional units. The lack of direction in defining the geographical attributes of functional units was found to have a trickle-down impact in those LCA stages that related inventory flows to the impact assessment and interpretation phases. It seems clear that explicitly identifying the geographical attributes in the preceding LCA stages could provide a simplistic way of being able to check the validity of these transformations and understand the true nature of their impacts relative to other LCA results.

The final observation made in the critical review was that land use change is often discussed synonymously with changes in soil carbon. It is well understood that LUC is a broad term that includes a series of different changes over time and should not focus on one aspect or indicator alone, including soil carbon changes. There needs to be careful attention paid to how indicators and impacts are defined so that interpretation of results are not made on the basis of complex issues that have been reduced to single parameters of analysis. Ultimately this observation reiterates many of the other conclusions made from reviewing bioenergy LCAs which is that the multi-scalar reality of their product systems necessitates more explicit methodological direction in order to account for space-time differences implied by different scales of observation.

This review highlighted the challenges one can encounter when attempting to interpret the environmental impacts of bioenergy technologies from studies that have not explicitly considered the role that geography has played in the data and results they generate. As more data from many different locations and scales of observations becomes available, without better direction on how to manage and integrate data from varying geographies in LCA, inconsistent approaches to system boundary selection and data management approaches will persist. The multi-scalar reality of bioenergy LCAs necessitates more explicit methodological direction in order to appropriately and consistently account for space-time differences implied by different scales of observation. Different information is generated at different research scales; this means that the interpretation of data is scale-dependent and that one must be careful when synthesizing this data. Improving LCA and transparency of geographical attributes is ultimately about understanding that more detailed knowledge of the system and its processes can improve the interpretation of system impacts.

5.4 Use of modular LCA approach as a means to solve issue of scale and dynamism

The dynamic model framework developed and used to assess the potential environmental impacts of growing short-rotation coppice willow as a bioenergy feedstock in southwestern Ontario showed that deployment of such a feedstock could have a positive net effect on greenhouse gas emissions, net energy balances, and soil carbon balances. The modular approach illustrated the way in which results can better reflect the reality of small-scale processes when geographical attributes (spatial and temporal) of smaller-scale processes (relative to the functional unit and product system) are maintained. By structuring the LCA in such a way that processes can be calculated individually at multiple scales, geographic attributes are not compromised, and indeed can be used to facilitate better communication between the functional unit, inventory flows, and results, particularly if data has been aggregated to or from the site of interest.

The results of a modular approach were also better able to direct future data collection and LCAs in terms of which sub-processes are likely to contribute the most variation and should consider finer-scale inputs. For example, knowledge of the characteristics of willow clone variations is not yet expansive, and data is lacking on key elements including the upstream processes required to provide inputs to develop these clones. Inclusion of clone-specific data can have significant impacts on other processes such residue inputs to soils. There are also significant gaps in soil carbon data that are needed in order to validate future models, as the soil carbon compartment is very sensitive to changes in the system; soil carbon can have a large effect on the long-term environmental impacts of the feedstock production system. Having a platform that can integrate data from different scales not only enables what data is available to be maximized in LCA, but also allows for future site-specific studies and data to be considered and added on to existing LCA frameworks.

5.5 Future Implications & Work

The modular approach demonstrated here also showed how a common spatio-temporal platform to connect multiple modules can help to manage complex systems in LCA that may operate at multiple scales; this has implications for LCA outside of bioenergy and perhaps future sustainability challenges outside of LCA. The inability for data from multiple sources and over large areas to be integrated into decision-making approaches has been a challenge for some time. Applying the modular approach described here has future implications for addressing those challenges in an age where digital data is increasing and our capacity to store it is becoming more accessible.

5.5.1 Expanding Indicators and Impacts

The importance of soil carbon sequestration has environmental impacts and importance beyond the climate change agenda. Carbon sequestration in the form of biomass has significant importance for pH, drainage, health of the willow crop, and ability for nitrogen, phosphorous and other fertilizer compounds to remain in the soil and crops. As we start to conceptualize sustainability in the context of not just global, but local impacts on the environment, large-scale indicators such as climate change are increasingly ignoring the more nuanced environmental picture. Selective analysis of only some of the appropriate indicators only provides a partial picture of the environmental impact, and there is a need to include all forms of impact and pollution to all three areas of our environment (air, water, soil), and critical changes to ecosystem values such as biodiversity, soil carbon, and water quality. Sustainability indicators such as reducing green house gas emissions may represent one issue surrounding energy sustainability (climate change), but may lack detailed analyses of other areas as the context changes (land-use change, resource depletion, cost-benefit analyses).

Part of the challenge in applying some of these values has been that many of these indicators need to be observed at a variety of scales. The approach used in this dissertation, which involves defining the scale attributes (spatial and temporal) of input data and system boundary definitions, not only facilitates a platform to compare multi-scale processes and therefore multiple indicators, but more importantly creates a transparent system whereby the practitioner or peer-reviewer can determine the appropriateness of the way that geographical characteristics are being applied from one LCA stage to the next. This type of LCA database platform would also allow for inventory data from new and emerging studies to be organized as they are produced and would allow practitioners across a range of disciplines to coordinate efforts that might start to reduce gaps in environmental impact assessment data and valuations. These revelations have implications for future energy policy, particularly in upcoming discussions about carbon-cap and trade pricing and other expected changes to GHG emission policies. In order to accommodate these considerations, indicator sets within these types of models need to be adequately flexible so that the expectations and values of local stakeholders can be met, emphasizing the importance of finding answers through perceived landscapes. As a result, some of the specific modules within the life cycle tool that will be important to expand in the future include:

i. expanding on soil carbon and nitrogen empirical models to better evaluate nutrient dynamics and identify where additional field-work data is needed

ii. moving biomass productivity assessments away from large-scale averages towards using biomass yield models such as light-use efficiency or allometric approaches

iii. linking of important feedback processes between land management activities, biomass production, and soil dynamics such as irrigation, implications of chemical applications, or alternative cropping techniques

iv. adapting life cycle tool interfaces to better accommodate feedstock producer needs in order to move life cycle assessment away from solely academic practitioner uses

5.5.2 Emerging Small-Scale Agricultural Approaches

In the context of bioenergy feedstocks, it has been identified that large-scale aggregated data has a tendency to inaccurately reflect site-specific dynamics that may significantly influence environmental performance. However, this phenomenon is not solely a bioenergy issue as it has important implications for other emerging sustainable agriculture technologies and initiatives such as intercropping, greenhouse crop production, permaculture and food forests, roof-top gardens, CSA farms, organic farming, and community gardens which all depend on highly site-specific characteristics and land-management approaches. Assessing these sustainable agriculture initiatives requires an interdisciplinary approach that requires socio-economic and community engagement, environmental assessment and natural resource management, and technical understanding of physical and biochemical processes.

As emerging carbon cap and trade policies emerge, it will be imperative that many of these sustainable agriculture producers operating at commercial levels are able to accurately assess and identify the environmental performance of their products and activities in order to promote sustainable approaches and remain competitive selling their farm products. Some of the site-specific dynamics that LCA will be called upon to compare if it wishes to participate in the analyses of such systems may include:

i. short- and long-term changes in soil carbon and nutrients in terms of mass balance

ii. changes in the productivity and yields of biomass and food production on both a spatial and temporal basis

iii. environmental impacts of sustainable agriculture operations in terms of GHG emissions, eutrophication, biodiversity, acidification, and land-use change

iv. technical bottlenecks associated with technology and/or data

v. potential co-products and reduction in waste streams associated with circular economies that may improve operating costs and reduce impacts

vi. Region-specific transport routes and types to understand costs and impacts of transport and ways to coordinate transport and improve efficiency

5.5.3 Implications for Canadian Energy Policy

Energy policies on a whole are aimed at influencing the sources and supply of energy, the demand for energy by a number of users, and the environmental impacts of energy use. Given Canada's extensive geographic range but heterogeneous natural resource and population distributions, future sustainable energy policies are going to need to be able to consider a range of energy needs, options, and local politics. As momentum towards local sustainability increases, energy policy is becoming an increasingly more complex interregional and multi-scalar matter in terms of dealing with the ownership and maintenance of energy infrastructure, electricity grids, and the liberalization of some energy markets.

With no clear definition of what sustainable energy strategy is (Liming et al 2008), the current policy objectives being pursued by the Federal and Ontario governments are lacking a central vision of sustainability, potentially dis-empowering regional initiatives and inhibiting the success of policy objectives aimed at market stimulation. Without a common strategy for both bottom-up action in projects and top-down approaches to legislation, we risk creating policy for one intended purpose without understanding the effects on another. Training in the installation, operation, and maintenance of renewable technologies is still fairly limited across Canada and receives marginal government support (MFFC, 2009). Municipalities have raised concerns that the capacity for knowledge transfer at many community levels is missing or inadequate (Fraser et al 2006); they don't have a broad knowledge base about renewable energy deployment or impacts to local ecosystems, and need help from other levels of government, municipalities, associations, agencies or training institutions.

Within the energy sector in Ontario, there is a specific need by policy makers to understand how decisions affecting the use of agricultural land for new biofuel production might influence or compete with other areas of agriculture and the environment. As shown in this dissertation, on-farm, region-specific data are scarce for biomass crops in North America, thus the quality of data currently available for policy decision-making is limited. Emissions and impacts for each process and the intensity of the activities within those processes must be identified using the appropriate environmental indicators, so that on-farm quantification of required data is adequate and avoids overlooking certain small-scale impacts that should be included in the analysis. Understanding what stages are the most energy and environmentally intensive, can also provide for important analytical tools in the future, when looking back to understand what sections of the system boundaries, or at what times within those boundaries, the environmental and energy impacts can be improved.

As a result, it is recommended that a coordinated effort between different levels of government, agricultural producers, and impact assessment practitioners to create a geographical information database of relevant impact and process data for bioenergy systems in Ontario and Canada could help facilitate answering some of the challenges identified above. Each scale of government, land-ownership, or environment use a particular scale of observation to describe their experiences and keeping track of the relevance of that scale used to define characteristics of the entities that define that particular level is important to keep track of and compare. By breaking down the impacts into regular space and time intervals, a better assessment of the environmental impacts would be possible, particularly in a Canadian and North American agricultural perspective, where seasonal work and growing seasons can drastically alter environmental and energy consumption impacts across geographies.

My current work with OMAFRA and the University of Guelph identified several important gaps in data and information needed in order to create better provincial policies that could support bioenergy deployment and expand feedstock production. Similar gaps exist for other sustainable agriculture approaches that may exist at other scales, including municipal, institutional, or regional policies. While the scales of these projects may change, the information needed to inform decision-making remains similar such as identifying important ecosystem services to support, accurate evaluation of environmental performance, and linking producers, policy decision-makers, and environmental assessment experts in an integrated platform that can facilitate common concerns and modes of communication. Sustainability requires integration of multiple scales and disciplines and therefore requires adoption of accessible language and approaches.

5.6 Final Remarks

This work has explored the modeling of complex systems in LCA and will be useful to inform future life cycle work as well as related renewable energy impact assessments (such as multiple perspectives, feedback loops, and adaptive capacities). It is clear that there are inherent and constantly evolving methodological challenges to further define geographical dimensions within LCA. Land use decisions will always have some implications for land elsewhere, beyond the defined boundaries of the system in question; ecosystems dynamics don't obey the arbitrary limits of the systems under consideration. In terms of developing sound sustainable energy criteria, ecosystem realities have broad-reaching implications for future technology applications and policy makers in Canada. Ecosystem dynamics are dominated by a variety of integrated negative and positive feedback loops, where uncertainty and limited predictability are unavoidable, meaning models need to embrace the eco-social complexity and uncertainty that exists from one regional perspective to another, in order to accommodate the visions, values and energy needs at different spatiotemporal scales.

The choice of any particular criteria to organize system observations – for instance, arbitrary spatial or temporal limits - carries the risk of ignoring not just other scales but also the inter-scalar relationships that are critical to successful policies. The evaluation of environmental systems and the development of energy policy should attempt to recognize other scales of operation. Conceptualizing ecosystems with people in them, as opposed to separate from them, means that new ecosystem approaches to LCA must be able to incorporate social processes simultaneously with ecological ones.

There is a need for shared concepts between social and biophysical disciplines, focusing on the patterns and processes across scales to influence socio-ecological analysis and governance. Integrated geographical approaches offer a an emerging and exciting disciplinary platform from which these types of challenges could be discussed, particularly in bioenergy LCAs, and warrants further exploration and expansion of the conclusions and suggestions expressed in this thesis.

5.7 Literature Cited

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