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Multiple Criteria Decision Analysis for the Selection of a Land Use Impact Method for a Life Cycle Assessment of Switchgrass as a Bioenergy Feedstock in the Pee Dee Region of South Carolina

Joel Kohn

Clemson University, joel.lovro.kohn@gmail.com

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MULTIPLE CRITERIA DECISION ANALYSIS FOR THE SELECTION OF A LAND
USE IMPACT METHOD FOR A LIFE CYCLE ASSESSMENT OF SWITCHGRASS
AS A BIOENERGY FEEDSTOCK IN THE PEE DEE REGION OF SOUTH
CAROLINA

A Thesis
Presented to
the Graduate School of
Clemson University

In Partial Fulfillment
of the Requirements for the Degree
Master of Science
Environmental Engineering and Science

by
Joel Lovro Kohn
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Accepted by:
Shelie A. Miller, Committee Chair
Cindy M. Lee
Stephen J. Moysey

ABSTRACT

The interactions of a growing human population and increasing demand for food and energy have led to governmental and social pressures encouraging the adoption of biofuels as a substitute for fossil energy sources; however, several potential biofuel feedstocks can compete directly with food products for valuable land area. Life Cycle Assessment (LCA) is a tool that examines the environmental impacts of a product or process and can assist decision makers in the development of policy. The environmental impacts of land use are not well incorporated into LCA. As an emerging field, there is no consensus regarding the best methods by which to include these impacts, and as a result, many methods have been proposed. A literature review was conducted of methods proposed since 2000 to include land use impacts in LCA. After compiling a list of methods, Multiple Criteria Decision Analysis (MCDA) was used to select a method to apply to a case study: the production of switchgrass as a bioenergy feedstock in the Pee Dee region of South Carolina (SC). The methods proposed by Weidema and Lindeijer (2001) and Mila i Canals et al. (2007) emerged as promising methods. After application to a case study of switchgrass production for biofuel applications in the Pee Dee region of South Carolina, the impacts to carbon emissions emerged as significant, and tradeoffs between carbon sequestered by switchgrass growth and lost assimilative capacity due to land use change were examined.

DEDICATION

I dedicate this thesis to my friends, my family, and my wife. Without their help and encouragement, I would not have accomplished what I have thus far.

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I would like to acknowledge my advisor, Dr. Shelie A. Miller and my committee: Dr. Stephen J. Moysey and Dr. Cindy M. Lee for their critical review of my thesis. I would also like to acknowledge my research group. They have been instrumental in the development of this thesis through support, comments, suggestions, and simply offering sets of eyes and open minds.

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CHAPTER ONE

INTRODUCTION

BACKGROUND

Life Cycle Assessment (LCA) is a tool designed to take into account all of the environmental impacts in addition to the energy and material inputs required to make a certain product or process from resource extraction to ultimate disposal or recycling.

LCA consists of four phases as defined by ISO 14040 and 14042 (2006 a, b):

1. Goal and Scope
2. Inventory Analysis (LCI)
3. Impact Assessment (LCIA)
4. Interpretation

During the goal and scope definition phase, the LCA practitioner defines both the boundaries of the system under investigation and the purpose of the study (ISO. 2006a). It is also in this phase that the practitioner selects a functional unit, the basis for comparison throughout the study (Hertwich et al. 2002). Goal and scope definition is followed by LCI, in which the practitioner collects data related to the environmental stressors (interventions) for each stage of the product's life cycle. Data can come from the main producers associated with the life cycle processes or from LCA databases for standard inputs. In the LCI phase, allocation of interventions to functional units can be complex, especially in the case where life cycle processes produce more than one output.

The results of the LCI enter the LCIA phase where they are classified into the relevant impact categories (e.g. global warming and ecosystem impacts) and characterized into a small set of indicators. This is the calculation phase, and the numerical results are normalized, grouped, or weighted at the discretion of the practitioner in keeping with the goal and scope of the study (ISO. 2006b). LCIA is often performed with the aid of dedicated LCA software to ease data management and repetitive calculation. In the final phase, interpretation, the numerical results are critically evaluated, which also includes discussion of sensitivity to assumptions and data quality (Hertwich et al. 2002).

LCA is an iterative process, where each step affects the following step but may also require the revisiting of a previous step to refine it further (Hertwich et al. 2002, Lindeijer et al. 2002b). The utilization of land by humans, henceforth called land use, has long been acknowledged to impact the natural environment and future land use is expected to impact biodiversity more than any other environmental intervention (Watson et al. 2000).

Since the impacts of land use can be significant in the life cycle of a product, these impacts should be included in LCA. However, these impacts are not easy to include in LCA as land use impacts differ fundamentally from those impacts traditionally included in LCA. As such, a variety of methods have been proposed (Blonk et al. 1997, Müller-Wenk. 1998, Lindeijer. 2000a, Lindeijer. 2000b, Köllner. 2000, Michelsen. 2008) to address the problem with no accepted best practice yet (Bare. 2009, Finnveden et al. 2009). The proposed methods differ significantly in the approaches they take to assessing land use impacts in LCA.

One particularly land-intensive sector is that of renewable energy (Fthenakis and Kim. 2009, Knoepfel. 1996, Lenzen and Munksgaard. 2002, Stoglehner. 2003, Dones and Frischknecht. 1998, Dornburg et al. 2003, Graham et al. 2000). Both biofuels and the renewable energy sector in general are poised for rapid growth as both societal pressures and governmental legislation are encouraging alternatives to fossil fuels, e.g. the Energy Independence and Security Act (U.S. House of Representatives. 2007). As biofuel production is increasingly implemented, the growth of feedstocks for those biofuels can compete with food crop production for land space. Tilman et al. (2009) assert that human society cannot afford the cost of biofuels done wrong (Tilman et al. 2009).

With a growing need to assess the impacts of land use, and the variety of options in implementing those impacts in land use, the LCA practitioner is left with a bit of a quandary: with no best practice and a variety of options, selecting a method to apply can be daunting. Multiple Criteria Decision Analysis (MCDA) is a class of tools that could potentially help the LCA practitioner.

MCDA consists of a variety of tools that are designed to select an option based upon a series of criteria that are not necessarily directly comparable (Munier. 2004). While MCDA cannot make the value judgments necessary for such a comparison to be objective, MCDA does make those value judgments explicit and transparent (Belton and Stewart. 2002).

PROBLEM STATEMENT AND OBJECTIVES

The problem of interest in this study is that despite the call for land use impacts to be actively included in LCA and tested in case studies (Baitz. 2007), the impacts of land use are not commonly incorporated into LCA (Bare. 2009, Finnveden et al. 2009). The objectives of this study are to compare and contrast potential land use impact assessment methods. The growth of switchgrass as a bioenergy feedstock in the Pee Dee region of South Carolina will be used as a representative case study to inform analysis. The final objective of this research project is to encourage discussion of impact methods in the larger LCA community to advance the inclusion of land use impacts in LCA.

This project employed the following process to achieve these goals:

1. Conduct an extensive literature survey to compile land use impact assessment methods.
2. Classify methods based on impact assessment approaches.
3. Compare and contrast methods based on number and type of characterization factors, number and type of impact indicators, data requirements, calculation methods, and whether the method uses valuation.
4. Apply MCDA to list of methods.
 - Generate list of criteria.
 - Assign scores to methods for each criterion.
 - Determine overall values for each method
5. Apply selected method or methods to case study.
6. Publish research as two separate papers in appropriate journals.

The method of MCDA used in this study is a value function method, also called grid analysis, and it is an additive method of MCDA; the overall value of a method is based on the sum of its scores for each criterion (Belton and Stewart. 2002).

The literature review, associated analysis, and the initial list of possible land use impact assessment methods for use in LCA are presented in Chapter 2. Chapter 3 contains the MCDA applied to a modified list of land use impact methods based on criteria adapted from literature sources and the results of the case study. Overall conclusions are presented in Chapter 4.

CHAPTER TWO

LITERATURE REVIEW

ABSTRACT

A review was performed of proposed methods to include the impacts of land use in Life Cycle Assessment (LCA). A number of authors have proposed different methods to include the impacts of land use since the early 2000s, but there is still no consensus regarding which to use. Several of the more recently proposed methods are expansions upon the authors' previous work and thus share similar elements with that previous work. One key point is the coarse spatial resolution of life cycle assessment, which often occurs on a larger scale than the impacts of land use, which are spatially dependent. The state of land before, during, and after the studied land-use is dependent on the specific location of the site; and although this state can be generalized to a certain extent, it is still at odds with the historically aspatial context of LCA. The allocation of land-use impacts over a chain of land uses is also a problem. Management decisions, such as conservation tillage practices and chemical application scheduling, can have a significant effect on the impacts of land use, yet few methods propose to address this. This review draws comparisons between proposed methods and also examines the frameworks for land-use proposed by the Society for Environmental Toxicology and Chemistry (SETAC) and the United Nations Environment Program (UNEP)-SETAC Life Cycle Initiative.

1. INTRODUCTION

Life cycle assessment (LCA) is an evolving tool designed to account for all of the environmental impacts associated with a product and its associated processes, from cradle to grave. Initially developed for industrial products, the application of LCA has become much more widespread. With the emergence of biofuels as a potential replacement technology for petroleum-based fuels, and the importance of food production to a growing global population, LCAs of agricultural and forestry products are becoming more prevalent (Michelsen. 2008, Haas et al. 2000, Weidema and Meeusen. 1999, Mattsson et al. 2000, Brentrup et al. 2004, Guinée et al. 2006, Antón et al. 2007, Miller et al. 2007). A key feature of agricultural and forestry production is the occupation of large amounts of land.

Land provides the surface to support all terrestrial life, and the functions of land within ecosystems include nutrient cycling and erosion resistance. Some land functions, like agricultural production, have environmental impacts that are already accounted in LCA (Mila i Canals et al. 2007a). For example, fertilizer runoff from farms contributes to eutrophication. This and other impacts (ozone formation, global warming potential, etc.) are already included in LCA and do not need to be included in a land use metric. However, many of the ecosystem aspects of land, like the existence of biodiversity, the continued ability for land to support life, and the primary productivity of land, are not currently accounted in LCA (Bare. 2009).

The issues associated with the use of land have recently come into further prominence with the development of plant-based alternatives for transportation fuel,

power generation, and commodity plastics and the desire to compare bio-based to fossil production (Mila i Canals et al. 2007a). Furthermore, the renewable energy sector includes energy production from wind and solar installations, both of which can require large land areas (Fthenakis and Kim. 2009, Lenzen and Munksgaard. 2002, Dones and Frischknecht. 1998, Schleisner. 2000). To make a comparison between petroleum-based products and plant-derived ones and to assess the impacts associated with renewable energy production within the context of LCA, the environmental impacts of land use should be taken into account (Lindeijer et al. 2002b, Mila i Canals et al. 2007a, Bauer et al. 2007)

Land use in the context of LCA is the human appropriation of land for a specific purpose. Examples of land use include agriculture, forestry, and urban development. Land use change refers to the changes in the physical coverings (trees, plants, buildings, etc.) required to prepare that area for a new use (Lindeijer et al. 2002b, Watson et al. 2000, Mila i Canals et al. 2007a). Indirect land use change is an economic consideration in that it is based on the concept that the global demand for a food crop will remain if arable land is used to produce a dedicated biofuel feedstock instead. Land elsewhere will thus be converted to cropland to make up for the decrease in food supply (Searchinger et al. 2008, Fargione et al. 2008, Kløverpris et al. 2008a, Kløverpris et al. 2008b).

The Energy Independence and Security Act of 2007 (U.S. House of Representatives. 2007) requires the inclusion of environmental impacts due to land use, land use change, and indirect land use change in the consideration of the life cycles of renewable fuels in the USA. The LCA community is thus being asked to include impacts

that are not well defined into their studies. In fact, the *direct* effects of land use and land use change are not yet incorporated into LCA, even though legislation requires the consideration of both direct and indirect land-use impacts in the life cycles of renewable fuels. There are a number of proposals intended to deal with this issue, and there is a UNEP-SETAC working group that is currently developing operational characterization factors for land use impacts on biodiversity and ecosystem services (Finnveden et al. 2009, Sonnemann and Valdivia. 2007, UNEP-SETAC Life Cycle Initiative. 2009).

There is significant debate in the area of including land use impacts in LCA, with differing opinions of topics ranging from the type of indicator chosen to the appropriateness of including land use in LCA. The release of the UNEP-SETAC Life Cycle Initiative (U-SLCI) framework (Mila i Canals et al. 2007a) for including land use in LCA generated several editorial responses, including one (Udo de Haes. 2006) that urged the LCA community to avoid trying to force aspects that do not fit into the tool, the so-called “Cinderella effect.” Another response to the framework urged the LCA community to gather inventory data and try methods out, and asserts that imperfect models are better than no models (Baitz. 2007). Despite the debate, it is clear that land use can have significant effects on the environment and should be included in LCA.

The impacts associated with land use include changes in biodiversity, soil organic carbon, land productivity, and buffering capacity and are not necessarily well-represented by linear relationships to the changes in how a piece of land is used (Michelsen. 2008, Arrhenius. 1921). The impacts of land use and land use change are significant (Lindeijer et al. 2002b, Mila i Canals et al. 2007a, Searchinger et al. 2008, Fargione et al. 2008,

Heijungs et al. 1997), as land use and land use change are predicted to be the primary drivers for the loss of biodiversity in the 21st century (Watson et al. 2000, Sala et al. 2000).

Within the context of LCA, land use and land-use change are closely related to the concepts of occupation and transformation, terms used by the SETAC LCIA guidelines (Lindeijer et al. 2002b) to describe the human interventions upon an area of land. Occupation refers to the use phase, during which the product or process under consideration operates. Transformation refers to the change phase, during which the surface is altered to suit a new use (Lindeijer et al. 2002b, Blonk et al. 1997, Mila i Canals et al. 2007a, Köllner and Scholz. 2007). Interventions (also environmental/anthropogenic interventions or elementary flows) refer to the physical elements that cross the system boundary into or out of the investigated system. Interventions also include changes to the physical environment such as clearing of trees and lowering the water table (Udo de Haes and Lindeijer. 2002).

This literature review presents methods proposed to include the environmental impacts of land use and land-use change in LCA. There have also been frameworks proposed, within which further methods could be developed (Lindeijer et al. 2002b, Mila i Canals et al. 2007a); however, the reviewed methods do not necessarily fit these frameworks. This is a review of methods proposed from 2000-2009 inclusive, as the last major review of the subject published in a journal occurred in 2000 (Lindeijer. 2000). However, two earlier, frequently referenced methods are also included. The review is limited to articles and reports published in English.

2. BACKGROUND AND GENERAL ISSUES OF LAND USE IN LCA

LCA is an evolving tool, which began as an attempt to assess industrial products, whose impacts could be effectively modeled independent of time and space (Hertwich et al. 2002, Finnveden et al. 2009, Cowell and Clift. 2000, Pennington et al. 2004); thus the resultant impacts could be applied globally. Table 2.1 shows a brief comparison between land use impacts and impacts traditionally included in LCA. In a traditional product LCA, the bulk of the impact categories are related to chemical emissions. Climate change, stratospheric ozone depletion, acidification, eutrophication, human toxicological effects, eco-toxicological effects, and photo-oxidant formation are all associated with chemical emissions; abiotic resources, biotic resources, and land use are associated with inputs (Heijungs et al. 1997). In addition to the site generic framework and the nature of the bulk of impact categories, LCA uses impact models based on linear relationships between the inputs and outputs associated with a product. Linear models adequately describe the conservation of mass aspects of processes in LCA.

Table 2.1: Comparison of land use impacts and traditional product LCA impacts

Issue	Land use impacts	Traditional Product LCA impacts
Nature of impacts	Related to physical changes in environment.	Most related to chemical releases.
Site specificity	Site specific, can be generalized based on larger regions.	Site generic. Methods developing towards greater specificity.
Linearity	Generally non-linear.	Emissions generally linearly related to functional units produced.
Allocation	Transformation difficult to allocate.	With linear relationships, additional functional units have unit impacts.

The inventory data and impacts resulting from land use interventions are largely site-specific, and although LCA methods exist that allow for spatial differentiation

(Finnveden and Nilsson. 2005), they are not commonly applied in case studies (Bare. 2009). While it is recognized that the impacts of land use upon the environment are significant, there is no consensus within the LCA community over the proper way to include those impacts (Lindeijer et al. 2002b, Bare. 2009, Finnveden et al. 2009, Mila i Canals et al. 2007a, Bauer et al. 2007, Udo de Haes. 2006). Individual authors tend to focus on a particular impact such as impacts upon biodiversity, life support functions, or soil properties in the methods they propose. The phases that comprise land use and the allocation of those impacts contribute to the difficulty in its incorporation into LCA.

Land use is commonly understood to consist of three phases (Müller-Wenk. 1998, Mila i Canals et al. 2007a, Köllner and Scholz. 2007), two of which are considered human interventions and the third which may or may not be considered an intervention. Transformation and occupation are the interventions; renaturalization or relaxation is the third phase and is considered an intervention by some authors (Lindeijer et al. 2002b, Müller-Wenk. 1998, Mila i Canals et al. 2007a). As stated previously, transformation corresponds to land-use change, while occupation corresponds to land-use. Renaturalization refers to the cessation of the occupation intervention and the subsequent slow return to a more natural state, which may or may not be facilitated by humans (Lindeijer et al. 2002a). The transformation process is essentially instantaneous in comparison to the occupation and renaturalization phases; the production of one additional functional unit of product does not necessarily require the transformation or occupation of one more unit of land area. This is fundamentally different from the measured chemical releases typically measured, where additional units of production lead

to corresponding quantities of emissions. Additionally, the allocation of the transformation impacts to products from later uses is difficult, as land itself is not consumed. It is possible that a virgin forest could be cleared for agricultural fields, and subsequently those fields could be converted to pasture, paved for roads, and/or built up into a suburban area or city.

Within LCA jargon is the term impact chain and the further terms that make up an impact chain. Impact chains refer to the environmental mechanisms between interventions and impacts on midpoints, endpoints, or areas of protection (AoPs). For example, emissions of chlorofluorocarbons and halons can further break down into Cl and Br ions, which destroy ozone. Based on the reactivity or lifetime of the parent compound in the atmosphere, the midpoint of this chain measures the ozone depletion potential of the parent compound. The destruction of ozone leads to increased UV radiation and leads to the endpoints of increased incidence of cataracts and skin cancer—which apply to the AoP human health—and to damage to crops, marine life, and exposed plastics. These last three endpoints apply to AoPs other than human health (Udo de Haes and Lindeijer. 2002).

Endpoints can be grouped together to form AoPs. The currently suggested AoPs are human health, natural environment, natural resources, and manmade environment (Finnveden et al. 2009, Udo de Haes and Lindeijer. 2002). Characterization factors are used to quantify the relationships between the interventions and the impact indicators, which can be tied to midpoints, like ozone degradation potential, or to endpoints like increased incidence of cancer or cataracts. The endpoints can be aggregated via a

weighting step into a single indicator for the AoP, such as disability adjusted life years (DALYs) in the case of human health impacts (Hertwich et al. 2002).

Some of these relationships cannot be fully quantified, especially past the midpoint stage. However, full quantification may not be necessary to achieve adequate modeling for land use (Lindeijer et al. 2002b). The levels (midpoint vs. endpoint) to which indicators may apply also add to the difficulty of including land use in LCA. While the LCA community has made progress in incorporating land use impacts into LCA, the attempts are still in their early stages of development (Bare. 2009).

3. LAND USE IMPACT APPROACHES

Since 2000 there have been 13 methods proposed to include the environmental impacts of land use in LCA. There have also been proposals for an overarching framework in which new methods for including land use can be developed. The framework is presented first, followed by the methods. The methods are broken up into categories based on the types of indicators proposed. Single-indicator methods are further split into those based on biodiversity, life support functions, and thermodynamics. The life support function category is subdivided into biomass production and soil quality. Multiple indicator methods are described in each category in which they have indicators.

3.1 Overarching framework

The UNEP-SETAC Life Cycle Initiative has published a proposal for a framework within which new methods of assessing land use impacts in LCA can be developed (Mila i Canals et al. 2007a). This framework draws from a previous

framework (Lindeijer et al. 2002b), and other prior work (Blonk et al. 1997, Lindeijer. 2000b, Lindeijer and Alferts. 2001). In addition to serving as a backbone for assessment method development, this framework (Mila i Canals et al. 2007a) is also designed to encourage discussion and debate regarding the inclusion of land use impacts in LCA.

Within the framework, land is understood to have a certain quality, the measurement of which is left to the LCA practitioner to determine; the authors do not suggest a preferred approach. They do suggest that the impacts of land use fall into several categories: existence of biodiversity, biotic production potential—including soil fertility, and ecological soil quality that includes other functions of soil.

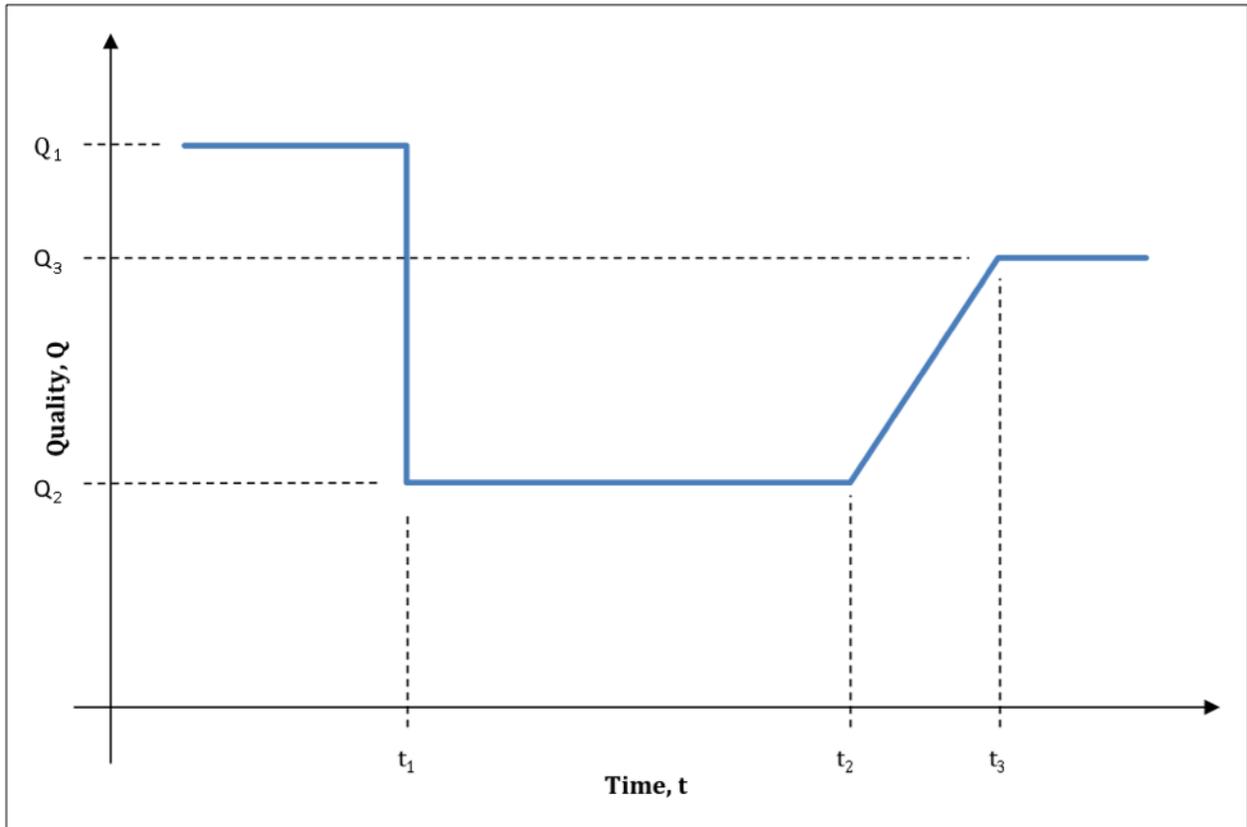


Figure 2.1: Changes in land quality associated with transformation, occupation, and relaxation. Q_3 is not necessarily less than Q_1 ; quality can improve. Adapted from (Lindeijer et al. 2002b).

As illustrated in Figure 2.1, the land in question begins at a certain level of ecosystem quality, Q_1 , which is degraded by the land transformation process at time, t_1 , to quality Q_2 . During the occupation process of duration t_2-t_1 , the quality as illustrated in Figure 1 remains constant at Q_2 . After cessation of the occupation process, the land relaxes back to a more natural state with final quality Q_3 during time t_3-t_2 . The relaxation time is not a function of the occupation; however, it is related to the processes that return the land to a state closer to its initial state. As illustrated, the land use has caused a net impact of Q_1-Q_3 . This illustrated situation is a simple one, but the same conceptual

framework applies to more complex situations where the occupation quality may not remain constant or may not be a linear function of time.

This simple situation also illustrates the issues of allocation of transformation impacts. The impact upon quality is not necessarily negative; it depends upon the reference situation, Q_1 in Figure 1, and the nature of the occupation and relaxation processes (Mila i Canals et al. 2007a, Köllner and Scholz. 2007). The Life Cycle Inventory (LCI) data needed for this framework are the area occupied, the duration of the occupation and transformation processes, and a quantitative description of the land quality for the entire situation. This framework considers the relaxation process to be a transformation process and thus an intervention.

The reference situation is an important consideration, and the authors suggest that a dynamic reference situation—the reference is taken as the situation that would have occurred if the studied system had not existed (Mila i Canals et al. 2007a)—be used as a basis for comparison, but ultimately the choice of reference depends on the indicators selected to represent the ecosystem quality (Lindeijer et al. 2002b). Mila i Canals et al. (2007a) also recognize the spatial dependence of ecosystem quality and impacts to that quality, and they suggest that spatial differentiation occur on the basis of biogeographical regions, which can be defined on a range of scales (Griffith et al. 2002).

The authors again stress the importance of the goal and scope of the study shaping every aspect that follows. Furthermore, the authors accomplish their goal of generating discussion as several letters to the editor (Baitz. 2007, Bauer et al. 2007, Udo de Haes. 2006) follow the publication of this framework.

3.2 Proposed impact assessment methods

The impact assessment methods proposed by various authors mainly differ in the indicators selected to represent the ecosystem quality, Q , from Figure 1. Table 2 summarizes the method proposals, which are divided based upon the impact categories to which they apply:

1. Biodiversity
2. Life support functions
 - a. Biomass production
 - b. Soil quality
3. Ecosystem thermodynamics

Table 2.2: Comparison of land use impact assessment methods

Method	(Number of) Characterization factors; type	(Number of) Quality indicators; type	Data requirements	Characterization factor calculation methods	Aggregation to single Quality score*
<i>Biodiversity</i>					
Muller-Wenk (1998)	(2) Marginal increase in threatened VPS ^a per km ² of land use; O ^b , T ^c	(2) Increase in threatened fraction of VPS; O,T	O & R ^d times, land use type, area	Swiss-specific marginal damage to VPS	No, weights suggested
Lindeijer (2000a)	(4) species area parameter initial, final, reference, actual based on VPS; O, T	(2) Ecosystem occupation, ecosystem change; O,T	Land use type, O time, ecoregion, area	Logarithmic species area relationship to derive indicator	No, but could use relaxation times
Kollner (2000)	(3) Species area parameter based on VPS; O,T, Regional	(2) Effect, O,T	Land use type, O time, area	Natural logarithmic species area relationship	No, relaxation times too variable
Mattsson et al. (2000)	(0)	(1) Species lost due to cultivated area	Species loss/area, cultivated area	n/a	n/a
Weidema and Lindeijer (2001)	(2) Quality difference; O,T	(2) Impact; O, T	Local quality-based on unaffected species; O, R time	Regional quality, product of 3 factors	yes
Brentrup et al. (2002)	(1) Naturalness degradation potential; O	(1) Naturalness Degradation Indicator; O	Land use type, area	Naturalness degradation potential assigned to 11 land use types	n/a
Kollner and Scholz (2007)	(4) species area parameters; O, T, R, baseline	(4) Damages; O, T, R, baseline	T, O, R times; land use type before, during, and after use	Linear species area relationship	yes
Michelsen (2008)	(1) Quality-based on 3 factors; O	(1) Impact; O	Ecosystem scarcity, eco-vulnerability, conditions for maintained biodiversity; O, R times	Quality, product of 3 factors	n/a
<i>Biomass</i>					
Blonk et al. (1997)	(1) fNPP ^e ; O	(1) Ecosystem damage; O	fNPP during O and baseline; O time; Area occupied	Difference in fNPP	n/a
Lindeijer (2000a)	(2) fNPP; O, T	(2) Ecosystem occupation, Ecosystem change; O, T	fNPP initial, final, baseline, actual; O time; Area occupied	Difference in fNPP	no
Weidema and Lindeijer (2001)	(2) Quality difference; O,T	(2) Impact; O,T	NPP ^f initial, final; O time, R time; Area occupied	Difference in NPP	yes
<i>Soil Quality</i>					
Cowell and Clift (2000)	(0)	(3) Static soil reserve life, SOM ^g indicator, soil compaction indicator	Global soil reserves, global soil loss rate; mass of SOM added to soil; weight of vehicle, field time, area	n/a	no
Mattsson et al. (2000)	(0)	(7) Δ (soil factor); erosion, hydrology, SOM, etc.	System and reference values for each soil factor; area occupied	n/a	no
Mila i Canals et al. (2007b)	(2) Life support functions; O, R	(2) SOM; O, R;	SOM system and baseline, SOM relaxation and SOM initial or final; O time, R time; area occupied	SOM measurements, models, or literature.	yes

<i>Ecosystem Thermo.</i>					
Wagendorp et al. (2006)	(0)	(3) Ecosystem surface temperature, Thermal response number, Solar exergy dissipation; not specified	Extensive remote sensing data	n/a	n/a
Dewulf et al. (2007)	(22) Land use exergy factors; O	(1) Cumulative exergy extraction from the environment; O	Area occupied, land use types	Calculated from energy denied the lowest trophic level	yes
^a Vascular plant species ^b Occupation ^c Transformation ^d Relaxation ^e free Net Primary Production = net primary production-human consumption ^f Net Primary Production ^g Soil Organic Matter *The implication here is multiple indicators are aggregated within the method, and does not necessarily reflect applicability to a subsequent valuation step.					

3.2.1 Impacts on biodiversity

Land use and land-use change are expected to be the key drivers of the loss of biodiversity in the 21st century (Sala et al. 2000). Eight of the reviewed methods focus on biodiversity impacts, which are frequently treated by using vascular plant species as a proxy for total species diversity. Müller-Wenk (1998), Lindeijer (2000a), and Köllner (Köllner. 2000, Köllner and Scholz. 2007, Köllner and Scholz. 2008) primarily develop their Characterization Factors (CFs), which link inventory to impacts, based on vascular plant species, and all cite references supporting that decision.

The individual CFs vary: the CF of Müller-Wenk (1998) is based on the ratio of threatened plant species to total plant species, while the CFs of Lindeijer (2000a) and Köllner (2000) are based on vascular plant species richness. Inherent in that treatment is the issue of non-linearity: as the observation area increases, the observed number of species experiences exponential growth (Arrhenius. 1921). Lindeijer (2000a) chooses a constant reference to make his CFs linear, while Köllner (2000) chooses a model more

similar to Arrhenius (1921) and does not choose a single reference state. Köllner and Scholz (2007 & 2008) find a good fit to their plant species datasets with a linear model. Each of the vascular plant species methods mentioned used data for local regions to generate CFs, thus applicability outside of Europe, or even other countries within Europe, may be limited.

Müller-Wenk (1998), Lindeijer (2000a), and Köllner (2000) do not aggregate their occupation and transformation indicators into a single quality score. Data availability and quality are cited as hurdles to aggregation (Lindeijer. 2000a, Köllner. 2000). Köllner and Scholz (2007 & 2008) give the transformation process a duration, rather than treating it as an instantaneous change. They also explicitly include the relaxation times and the relaxation process as an intervention, which simplifies aggregation to a sum. Köllner and Scholz. (2007 & 2008) also present CFs based on moss species and mollusk species to investigate the relationship between plant biodiversity and other species diversity. The four later publications (Lindeijer. 2000a, Köllner. 2000, Köllner and Scholz. 2007, Köllner and Scholz. 2008) all incorporate methods to account for differences between regions.

The methods proposed by Weidema and Lindeijer (2001) and Michelsen (2008) share some common elements. Weidema and Lindeijer (2001) propose three quality factors that are aggregated to a single indicator: species richness, ecosystem scarcity, and ecosystem vulnerability; Michelsen (2008) also proposes three quality factors: ecosystem scarcity, ecosystem vulnerability, and conditions for maintained biodiversity that are also aggregated to a single score. The species richness indicator of Weidema and Lindeijer

(2001) is based on vascular plant species per area and is normalized based on factors dependent on biomes. Michelsen (2008) modifies the ecosystem scarcity factor of Weidema and Lindeijer (2001) to reflect availability of data for ecoregions (Olson et al. 2001a)—finer resolution than biomes (Michelsen. 2008).

Ecosystem vulnerability is related to species area curves, similar to previously described methods (Lindeijer et al. 2002b, Köllner and Scholz. 2007, Köllner and Scholz. 2008), but it is based on the area of ecosystem area rather than the number of species found in it (Michelsen. 2008). The third quality factor of Michelsen (2008), conditions for maintained biodiversity, is an aggregation of sub-indicators known to influence biodiversity. These sub-indicators are assigned scores based on qualitative assessment. The sub-indicators are site-dependent/region-dependent and are based on expert knowledge, making maintained biodiversity difficult to apply outside of the studied regions. In the accompanying case study, data are not available to use ecosystem scarcity and ecosystem vulnerability as proposed, so the author suggests some simplifications based on ecoregion data and qualitative description (Michelsen. 2008).

Not all methods for impacts on biodiversity choose plant species as an indicator. Mattsson et al. (2000) propose a simple indicator: species lost as a result of crop cultivation. The authors and choose different species, including animal species, in the different locations under investigation. This indicator is part of a multi-indicator method, but the rest of the indicators are for impacts on soil quality. This method requires site-specific data and the included case studies use different species as indicators, which may not be directly comparable. This method also includes a qualitative description of

aesthetic value based on Swedish landscape preferences, but this analysis was not extended to the two case studies outside Sweden (Mattsson et al. 2000).

The method of Brentrup et al. (2002) involves a qualitative description of the naturalness, or hemeroby, of an area of land. The method assigns hemeroby values to 11 classes of land use based on qualitative descriptions, thus requiring only area occupied and land use type as inventory data, provided the land use types studied match with those proposed (Brentrup et al. 2002)

3.2.2 Impacts on biomass production

A second major category of land-use impacts is the impact upon biomass production (Lindeijer et al. 2002b, Lindeijer. 2000b, Mila i Canals et al. 2007a). The production of biomass, that is the fixation of carbon, is both one of the life support functions performed by a land and the main function of soil that humans use (Udo de Haes and Lindeijer. 2002). Three of the method proposals reviewed included significant discussion of production or productivity as an indicator for the impacts of land use in LCA: Blonk et al. (1997), Lindeijer (2000a), and Weidema and Lindeijer (2001). The methods of Blonk et al. (1997) and Lindeijer (2000a) are based on free Net Primary Production (fNPP), while that of Weidema and Lindeijer (2001) is based on Net Primary Production (NPP). NPP is the net carbon uptake of the system; fNPP is the portion left after human harvest. NPP is considered a proxy for nutrient cycling (Weidema and Lindeijer. 2001). fNPP is suggested as an indicator because it represents the amount of biomass available for nature to use (Blonk et al. 1997, Lindeijer. 2000a).

The method of Lindeijer (2000a) is an extension of the work of Blonk et al. (1997), and their treatments of productivity are essentially identical. Lindeijer (2000a) also includes a biodiversity indicator discussed in Section 3.2.1. Both methods express the quality change due to land use to be the difference in fNPP caused by that land use. While Blonk et al. (1997) do not treat transformation impacts; Lindeijer (2000a) does, but the aggregates the two into a single indicator. Similar to the treatment of Lindeijer (2000a), Weidema and Lindeijer (2001) use the change in NPP as a result of the studied land use as the indicator of productivity.

3.2.3 Impacts on soil quality

The production of biomass is not the only soil function of interest to humans, and the physical and chemical properties of the soil, as opposed to biomass production as a proxy, have also been proposed as an indicator to assess the impacts of land use in LCA. Three of the reviewed methods focused on soil quality indicators (Cowell and Clift, 2000, Mattsson et al. 2000, Mila i Canals et al. 2007). Cowell and Clift (2000) and Mattsson et al. (2000) use methods developed specifically for the case studies examined; they do not develop CFs that would facilitate application to other case studies. Cowell and Clift (2000) proposed three indicators: quantity of soil, organic matter, and soil compaction for use in LCA. Mattsson et al. (2000) proposed seven indicators: soil erosion, hydrology, soil organic matter (SOM), soil structure, soil pH, heavy metal accumulation, and potassium/phosphorus content for use in LCA, with additional biodiversity and landscape value indicators discussed in Section 3.2.3. The model of Cowell and Clift (2000) is recursive and attempts to take into account spatial and

temporal issues associated with management practices. Mattsson et al. (2000) are unable to implement all seven indicators in case studies due to limited data availability. Neither author addresses the difference between occupation and transformation, even though the description of biodiversity indicator of Mattsson et al. (2000) seems to describe an impact from transformation.

The three indicators of Cowell and Clift (2000) and the seven indicators of Mattsson et al. (2000) for soil quality both included SOM. SOM is a key soil quality indicator and may better represent situations where intensive management yields high biomass productivity at the expense of the soil (Mila i Canals et al. 2007b). SOM is proposed as a single indicator by Mila i Canals et al. (2007b) as a means to assess the land use impacts on life support functions. Using SOM data from direct measurements, models, or literature (preferential order) Mila i Canals et al. (2007b) propose to quantify land use impacts for occupation and relaxation, where the net change after relaxation represents the impact due to the land transformation. The authors also note that SOM alone fails to address biodiversity issues (Mila i Canals et al. 2007b).

3.2.4 Impacts on ecosystem thermodynamics

Similar to the impacts of land use on biomass, impacts on ecosystem thermodynamics generally refer to energy that is denied to the environment or changes in the ability of the ecosystem to dissipate energy as a result of land use. Two of the reviewed methods focus on indicators related to ecosystem thermodynamics: Wagendorp et al. (2006) and Dewulf et al. (2007). Their approaches differ significantly. Wagendorp et al. (2006) propose to establish a firmer theoretical basis for the ecosystem impacts due

to land use, and suggest three potential indicators based on remote sensing data: surface temperature, thermal response number, and solar exergy dissipation, where exergy is useful energy, i.e. energy subtracted of its entropic content. This method is not fully developed yet. To calculate these factors, the LCA practitioner would need remote sensing data for the specific site under study. The authors note several issues with remote sensing data, including edge effects and topography, which contribute to the difficulty in applying this method in practice (Wagendorp et al. 2006).

While still based on ecosystem thermodynamics, Dewulf et al. (2007) treat land use as a resource extraction and quantify the energy denied the ecosystem due to occupation of land. The authors propose to treat all resources in this fashion, and calculate CFs for the 184 reference flows found in the ecoinvent LCA database. For land use, the CFs are calculated from the energy denied the lowest trophic level, primary production. Of the calculated CFs, 22 are land occupation categories; land transformation is not considered as it is stated to not deprive the ecosystem of any exergy. The practitioner need only know the type of land use studied, the area, and the time of occupation in order to calculate the indicator (Dewulf et al. 2007).

4. DISCUSSION

While all of the methods reviewed are designed to incorporate the environmental impacts of land use into LCA, the approaches the various authors take can be quite different, both from a theoretical perspective, and the ease by which a method can be

applied to new case studies. The degree of fit within the proposed framework (Mila i Canals et al. 2007a) also varies.

It is interesting to note the evolution of thought associated with the incorporation of land use into LCA. Some key elements of earlier methods (Blonk et al. 1997), especially the chart of ecosystem quality vs. time, the use of some form of primary production, and the reduction of species and area to a linear relationship have persisted into the overarching framework and methods proposed in more recent publications (Lindeijer et al. 2002b, Mila i Canals et al. 2007a, Köllner and Scholz. 2007, Köllner and Scholz. 2008). There seems to be a fairly limited community that is actively involved in advancing the incorporation of land use impacts into LCA, with several individuals appearing regularly in publications. It may not be surprising then that the more recent method proposals include significant contribution from key individuals.

Even with the relatively small, active group, there is still no consensus regarding the proper way to include land use impacts in LCA. Several letters to the editor (Baitz. 2007, Bauer et al. 2007, Udo de Haes. 2006) with dissenting opinions were published in the wake of the UNEP-SETAC framework (Mila i Canals et al. 2007a). Opinions in the responses range from the suggesting that land use and LCA are mutually exclusive to suggesting that only by repeated method trials will the field advance. More recent reviews of LCA developments (Bare. 2009, Finnveden et al. 2009) continue to assert that the incorporation of land use impacts has a significant way to go before it will be operational on a large scale.

One aspect of current approaches that hinders incorporation is the connection between the indicators chosen and the impact modeled. Biodiversity indicators attempt to measure species richness as a function of area directly, and while authors use indicators that are linear with respect to area occupied in LCA calculations (Blonk et al. 1997, Lindeijer. 2000a, Köllner. 2000, Köllner and Scholz. 2007, Köllner and Scholz. 2008), the impact on the environment—loss of biodiversity—is not linear.

Single indicators may lack the comprehensiveness of multiple indicator methods (Mila i Canals et al. 2007b), while the multi-indicator methods have a greater chance to run into data availability issues (Michelsen. 2008, Mattsson et al. 2000). The basis upon which many biodiversity indicators lie, vascular plant species, is also debated. Michelsen (2008) rejects the use of vascular plant species as not representative of biodiversity as a whole, citing literature to support the position. At the same time, the indicators proposed instead of species richness are not used due to data limitations, but some guidance on alternate indicators is given (Michelsen. 2008).

The limited availability of data continues to fetter the use of land use impacts in LCA case studies. While some authors have calculated reference values that allow the LCA practitioner to use proposed methods based on a few more commonly available pieces of data, the reference values are generally developed in relation to a specific country or region and are based on qualitative descriptions of land use classes (Müller-Wenk. 1998 ,Köllner and Scholz. 2008, Dewulf et al. 2007). This makes application of the various methods to regions other than the ones in which they were developed

questionable. The practitioner is left to wonder if one can safely use the techniques and values reported in literature.

This data dearth is reflected in the lack of case studies that incorporate land use impacts (Bare. 2009, Finnveden et al. 2009). Not all of the method proposals were accompanied by full case studies; many included only simple examples. Conflicting recommendations further add to the difficulty. In his editorial response to the UNEP-SETAC LC Initiative framework, Baitz (2007) suggests that case studies should be an integral part of method development; by actively testing methods, progress will occur more rapidly. Alternatively, Guinee (2006) recommends that in cases of conservation agriculture, land use impacts should be limited to the inventory result of the product of time and area occupied. Case studies demonstrating method use are not very common (Bare. 2009), and they only accompany method proposals on a limited basis (Michelsen. 2008, Mattsson et al. 2000, Cowell and Clift. 2000). The accompaniment of method proposals by a worked case study instead of an example calculation could help the practitioner community move forward in applying methods to their case studies.

5. CONCLUSIONS AND RECOMMENDATIONS

Life Cycle Assessment is an evolving tool; as new methods and techniques are developed, the established ones must be examined to ensure the continued applicability of the tool as a whole. Likewise, new methods should be scrutinized to make sure that they continue to develop LCA in a way that is consistent with the goals of LCA: to examine the environmental tradeoffs associated with various products and processes, and

to help inform product development and public policy, ultimately moving society towards sustainability.

The UNEP-SETAC Life Cycle Initiative phase 2 work regarding land use impacts in LCA is scheduled for completion in May 2010 (UNEP-SETAC Life Cycle Initiative, 2009). Ideally these recommendations will represent a consensus view and will move the incorporation of land use impacts into active practice of LCA forward significantly. However, given past debate, future lack of consensus is a continued possibility.

The purpose of this review is to stimulate discussion of land use impact methods in anticipation of the UNEP-SETAC LC Initiative recommendations. The author also wishes to encourage scrutiny and thorough examination of new methods to continue the growing acceptance of LCA as an assessment tool, and to call attention to the similarities and differences in new methods. Many of the methods examined in this review at first seemed to share few features; however, after closer inspection, many shared significant similarities: approaches to non-linear species/area relationships, key indicators, and a consistent framework. An evolution of thought assessment methods was also observed as more complete datasets and simply more data for indicators became available.

This review also calls into attention the importance of the practitioner in determining the goal and scope of the study, as those decisions significantly affect the choices of indicators for a given study (Mila i Canals et al. 2007a), which has been a key aspect of LCA's flexibility and a possible liability since its inception. Future work in the area could include greater inclusion of land use impacts in case studies, especially those associated with land-intensive sectors and in bio-based vs. petrol-based fuel comparisons.

CHAPTER THREE

MULTIPLE CRITERIA DECISION ANALYSIS

ABSTRACT

The environmental impacts of land use are not well incorporated into Life Cycle Assessment (LCA), yet these impacts are of particular importance to agricultural products and other products of land-intensive activities, such as forestry. As bio-based products emerge to replace those dependent on petroleum, the land use impacts from those products gain importance in LCA. A number of methods exist to incorporate these impacts into LCA; however, there is not yet a method considered best practice. The aim of this study is to use Multiple Criteria Decision Analysis (MCDA) to identify a practical method to analyze the impacts of land use associated with the production of switchgrass as a bioenergy feedstock in the Pee Dee region of South Carolina (SC). After identifying several potential land use impact methods from a review of literature, the author applied an MCDA technique to select a method to apply to the case study. Two methods emerged from the MCDA process: Weidema and Lindeijer (2001) and Mila i Canals et al. (2007b). These methods were applied to the case study and the impacts associated with converting all productive cropland and Conservation Reserve Program land in the region to switchgrass production were calculated.

1. BACKGROUND, AIM, AND SCOPE

The Pee Dee region of South Carolina consists of seven counties: Darlington, Dillon, Florence, Lee, Marion, Marlboro, and Williamsburg in the northeast of the state where agriculture is significant (U.S. Department of Agriculture, National Agricultural Statistics Service. 2009). Switchgrass, a perennial prairie grass, is under investigation as a potential cash crop for the region. As a bioenergy feedstock, switchgrass can be processed into ethanol or co-fired with coal in power plants (Ma et al. 2000, Varvel et al. 2008). The impacts of bio-based production can be significant, especially upon biodiversity (Mila i Canals et al. 2007a, Sala et al. 2000).

The assessment of land use impacts is not yet well incorporated into LCA. There is debate and discussion surrounding this problem, but there is not yet a consensus regarding the proper way to account for these impacts. Two documents try to establish a framework within which method developers can work (Lindeijer et al. 2002b, Mila i Canals et al. 2007a), with the latter building on the former. Various authors debate the aspects of land use impacts in LCA as editorials following the Mila i Canals (2007a) publication (Baitz. 2007, Bauer et al. 2007, Udo de Haes. 2006).

Multiple methods exist that attempt to address the issue of land use in LCA (Blonk et al. 1997, Lindeijer. 2000a, Köllner. 2000, Michelsen. 2008, Mattsson et al. 2000, Köllner and Scholz. 2007, Mila i Canals et al. 2007b, Dewulf et al. 2008), and recent reviews of LCA (Finnveden et al. 2009, Bare. 2009) still identify the subject as having room to grow. The aim of this study is to apply a simple method of MCDA to the problem of assessing land use impacts in LCA. One of the strengths of MCDA is the

ability to make incommensurable factors comparable through value judgments (Belton and Stewart. 2002). The variety of methods proposed to include land use impacts in LCA make direct comparison difficult, making the situation a prime candidate for MCDA.

The result of the MCDA process will be a method or methods to take land use impacts into account. To illustrate the methods and provide insight into the analysis, a screening-level study of the land use impacts of switchgrass production in the Pee Dee region of SC. This case study will examine the impacts under the scenario that all productive cropland and degraded cropland is put into production. This includes the conversion of lands currently held in Conservation Reserve Program (CRP) back to active production to represent the largest amount of land that could be put into production without conversion of non-farmland. CRP lands can include a variety of vegetation covers ranging from degraded land to early-succession forests (Barbarika. 2009).

This case study is designed to give some insight into the tradeoffs associated with the land use impacts of the production of switchgrass. The functional unit is 1000 kg of dry biomass, as switchgrass can feed more than one product stream. The impact categories of biodiversity, ecological soil quality, and biotic production, suggested by Mila i Canals et al. (2007a), are of particular importance. SC has substantial biological and ecological diversity (Griffith et al. 2002), and the long-term ecological soil quality and biotic production potential are important for the continued production in an agricultural region.

2. GENERAL DESCRIPTION OF MCDA

Multiple Criteria Decision Analysis (MCDA) consists of a variety of tools that attempt to make explicit and transparent the various factors that go into all decisions. MCDA does not make those quality judgments any less subjective; it simply makes them explicit. MCDA can also serve as a means to encourage thorough examination of personal preferences in the process (Munier. 2004, Belton and Stewart. 2002).

The method chosen for this study is a utility function method, also called grid analysis, where the value of an option is defined as

$$U_a = \sum_{i=1}^n w_i s_{a,i} \quad (1)$$

Where U_a is the overall utility of alternative a , $s_{a,i}$ is the score of alternative a for criterion i , and w_i is the weight of criterion i , which reflects its relative importance compared to the other criteria. This is one of the simpler MCDA methods, as the value is based on the sum of factors. One method to determine the w_i values is via a series of paired comparisons, which forces a preference between each criterion. The scores, $s_{a,i}$, is assigned based on a qualitative value rubric. The utilities, U_a , allow comparison between alternatives. The alternative with the largest utility is the most preferable option for the given criteria and weighting scheme. It is important to note that the most preferable option is heavily dependent upon the chosen criteria, scores assigned for those criteria ($s_{a,i}$), and weights (w_i), and as each is based on qualitative judgments, the results can contain significant bias (Belton and Stewart. 2002).

3. METHODS

A review of English language publications identifies potential methods to include the impacts of land use in LCA. This study focuses on methods published since 2000 and ultimately identifies ten methods to serve as the options for MCDA (Lindeijer. 2000a, Michelsen. 2008, Mattsson et al. 2000, Mila i Canals et al. 2007b, Köllner and Scholz. 2007, Cowell and Clift. 2000, Weidema and Lindeijer. 2001, Wagendorp et al. 2006, Dewulf et al. 2007, Brentrup et al. 2002).

The impacts of land use can be examined within frameworks other than LCA. There exists a class of methods known variously as ecosystem services or natural capital valuation methods (Costanza et al. 1997, de Groot. 1987, Tallis and Kareiva. 2005, Nelson et al. 2009); these methods assign monetary value to the services that ecosystems provide to humans. These methods may be a viable way to assess land use impacts, and could make a valuable addition to LCA if the two frameworks can be reconciled. However this study excludes these ecosystem service methods as outside the scope of the study because they do not directly address land use impacts.

Blonk et al. (1997) identify seven criteria upon which the authors judge options for indicators to include the impacts of land use in LCA. While not nominally an MCDA process, the analysis and arguments the authors pose are superficially similar to the MCDA process. Their list served as a starting point for the judgment criteria for this study, though this study only uses five, listed below.

1. The method should provide an adequate description of the impact upon the natural environment.

2. There should be a linear relationship between intervention and impact, to allow extrapolation from one functional unit to many.
3. Objective/quantitative methods are preferred to subjective/qualitative ones.
4. The method should be able to include a valuation step to allow the possibility of aggregation at the end of the study.
5. The method should be feasible in practice; that is, the practitioner should be able to use available data to perform the study.

Blonk et al. (1997) include site independence and the applicability to all ecosystems and land use types as the two additional criteria. These two criteria were not included as they were unable to be quantified with a preference scale like the other five.

While the summaries of the criteria above serve as a starting point for the judgment criteria in use in this study, further explanation of the criteria helps to clarify the value judgments necessary for this MCDA. The method should apply to the impacts already identified as important for land intensive activities: biodiversity, ecological soil quality, and biotic production (Mila i Canals et al. 2007a). LCA is traditionally a linear tool (Blonk et al. 1997, Udo de Haes and Lindeijer. 2002), in which the production of an additional functional unit of product results in an additional unit of impact; methods to include land use impacts in LCA should therefore use linear relationships to translate inventory result into environmental impact. Objective or quantitative methods are preferred in LCA, as qualitative descriptions tend to get lost in the process of LCA (Lindeijer et al. 2002b). Valuation or aggregation of the results of the impact assessment into a single score is left to the discretion of the LCA practitioner (ISO. 2006b, Lindeijer

et al. 2002b); however, for public policy matters, legislators may desire a single score. Practical feasibility is considered the most important criterion for this study, as the goal is to use publicly available data to assess land use impacts. The practical criterion assumes that models developed for specific regions, e.g. Europe, either apply globally, or can be easily adapted to the local situation.

It is possible that no method will satisfy all criteria, and it is possible to conclude that land use impacts should therefore not be included in LCA. This has, in fact, been argued in letters to the editor of the International Journal of Life Cycle Assessment (Baitz. 2007, Bauer et al. 2007, Udo de Haes. 2006). However, given the projected importance of land use impacts (Sala et al. 2000, Watson et al. 2000), this conclusion is not acceptable.

The two additional criteria are excluded from this study because it is difficult to define a preference scale for them. Land use impacts are necessarily dependent on the location and the surrounding area (Blonk et al. 1997), but greater site-specificity increases data requirements. Some land use impact methods use classes of land use and eco-regions to represent local and regional situations and to serve as reference states (Michelsen. 2008, Köllner and Scholz. 2008). The claim of applying to every possible situation is questionable and difficult to value.

Using the five criteria detailed above, criterion weights were generated with the pairwise comparison method. Each criterion is compared on a 1-on-1 to the other criteria. For n criteria, $n*(n-1)$ comparisons are required to include every possible pairing. The preferred criterion in each pair receives a point. The raw score for each method was

normalized by dividing by the total possible points to generate the weight for each criterion. To ensure that all weights were non-zero, a sixth criterion was included that received no points. The comparison matrix that generated the criterion weights is presented in Table 3.1. In this study, the availability of data, the practical criterion (criterion 5 in Table 3.2) was preferred in each comparison. The rest of the comparisons were based on the authors' experience.

Table 3.1: Pairwise comparisons and total points for each criterion

Criterion (<i>i</i>)	Pairwise Comparisons													Row Sum	
1.Descriptive	1	1	1	0	1										4
2.Linear	0					0	1	0	1						2
3.Objective		0				1				1	0	1			3
4.Valuation			0				0			0			0	1	1
5.Practical				1				1			1		1		5
6.Dummy					0				0			0		0	0

Each method was then scored for each criterion based on the rubric presented in Table 3.2. The values for the scores were assigned based on the method proposal papers. For example, to generate the Descriptive criterion, three important impact categories were identified in Mila i Canals et al. (2007a). If in a method, the authors addressed none of those impact categories, the method scored zero for that criterion. Addressing one category scored 3, two categories scored 6, three categories scored 9, and more than three categories scored 10. Similar arguments were used to generate the rest of the rubric.

The practical criterion required the search for publicly available data, and an examination of data sources cited in published methods, while the rest of the criteria required critical examination of the texts in which the various authors proposed their methods. Arguments justifying the score of each method for each criterion can be found

in Appendix A. Because the weights and scores are themselves a series of value judgments, this added further subjectivity to the study and could be a significant source of bias.

Table 3.2: Score ($s_{a,i}$) assessment rubric. Fractions refer to the fraction of indicators that fit the criterion.

	Score ($s_{a,i}$)						
Criterion (i)	0	1	3	5	7	9	10
1. Descriptive	Does not relate to impacts of interest		Addresses one impact category		(6) Addresses two impact categories	Addresses all three impact categories	Addresses additional impact categories
2. Linear	No indicators linear	1/10 linear	3/10 linear	Half of indicators linear	7/10 linear	9/10 linear	All indicators linear
3. Objective	No indicators objective	1/10 objective	2/10 objective	Half of indicators objective	7/10 objective	9/10 objective	All indicators objective
4. Valuation	Valuation not possible		Valuation difficult: vastly different indicators	Valuation possible but not included in method	Result of impact assessment is small number of scores	Result of impact assessment is two scores	Result of impact assessment is single score
5. Practical	Data available for no indicators	1/10 data availability	3/10 data availability	Data available for half of indicators	7/10 data availability	9/10 data availability	Data available for all indicators

Using Equation 1, the weights (w_i) generated from Table 3.1, and the scores ($s_{a,i}$) generated from Table 3.2 the overall utility for each method is calculated using a spreadsheet application. This study also examined the sensitivity of the utilities for each method to changes in the weighting system, and to changes in the criterion scores. The resulting method(s) were applied to the case study and these results are discussed in Section 5.

4. RESULTS

The initial criterion weights were calculated by fifteen pairwise comparisons with the results shown in Table 3.3. Again, these values represent qualitative judgments based on the author's preferences and experience.

Table 3.3: Criterion weights from pairwise comparison

Criterion (<i>i</i>)	Weight (w_i)
1.Descriptive	.267
2.Linear	.200
3.Objective	.133
4.Valuation	.067
5.Practical	.333

Table 3.4 presents the results of subjecting each method to the scoring rubric in Table 3.2. While the rubric was broken down in a quantitative fashion for each criterion, these values are also subjective.

Table 3.4: Scores ($s_{a,i}$) for each criterion and method.

Method (<i>a</i>)	Criterion (<i>i</i>)				
	1. Descriptive	2. Linear	3. Objective	4. Valuation	5. Practical
Lindeijer et al. (2000a)	6	10	10	5	7
Weidema and Lindeijer (2001)	6	10	10	9	10
Brentrup et al. (2002)	3	10	0	10	9
Köllner and Scholz (2007)	3	10	10	10	9
Michelsen (2008)	3	10	3	10	7
Mattsson et al. (2000)	10	3	9	3	2
Cowell et al. (2000)	10	3	10	3	2
Mila i Canals et al. (2007b)	6	10	10	10	9
Wagendorp et al. (2006)	3	10	10	10	1
Dewulf et al. (2007)	3	10	10	10	10

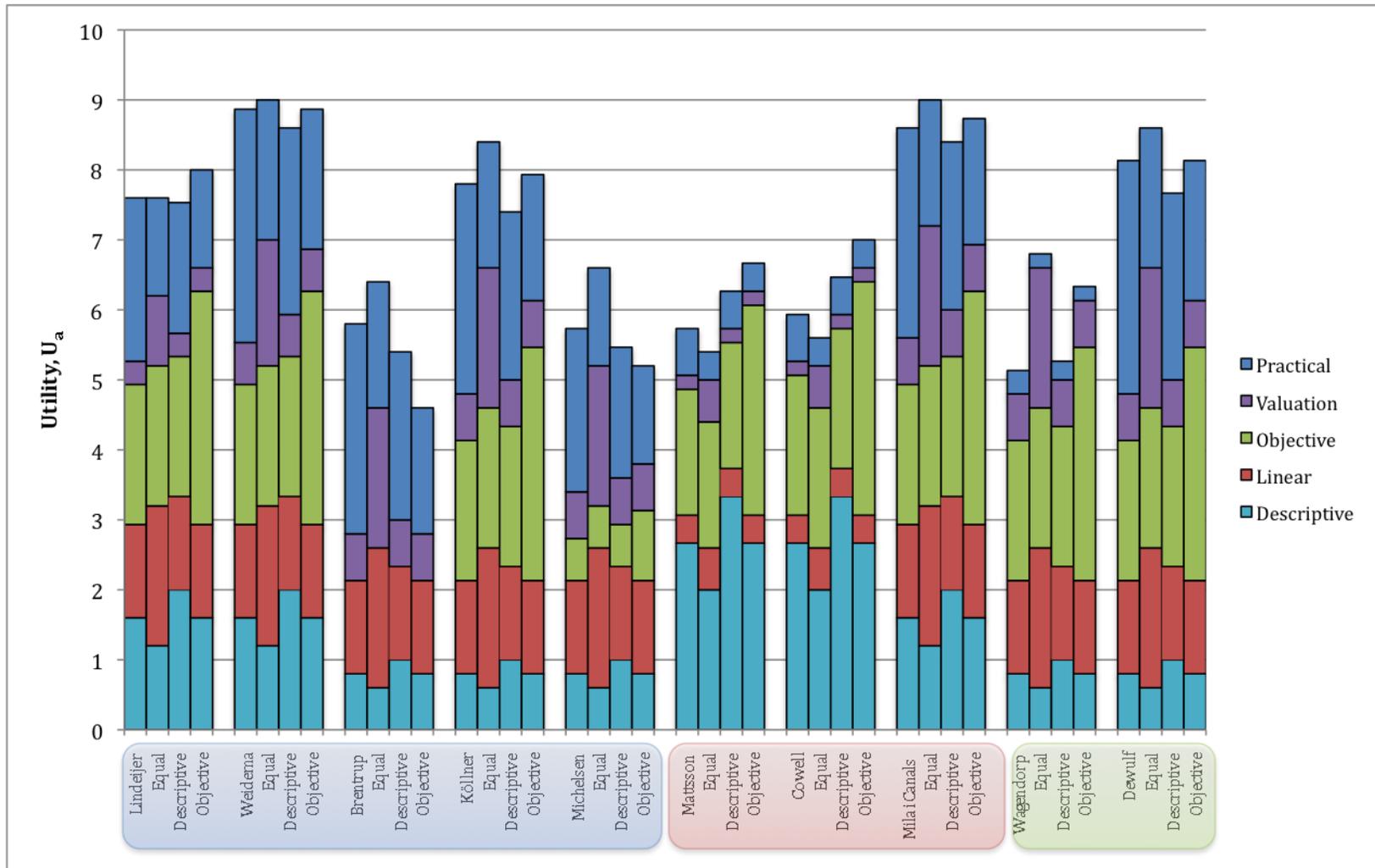


Figure 3.1: Results of overall value and sensitivity to weights for each method. The blue, red, and green accents denote biomass/biodiversity, soil, and thermodynamics based methods respectively.

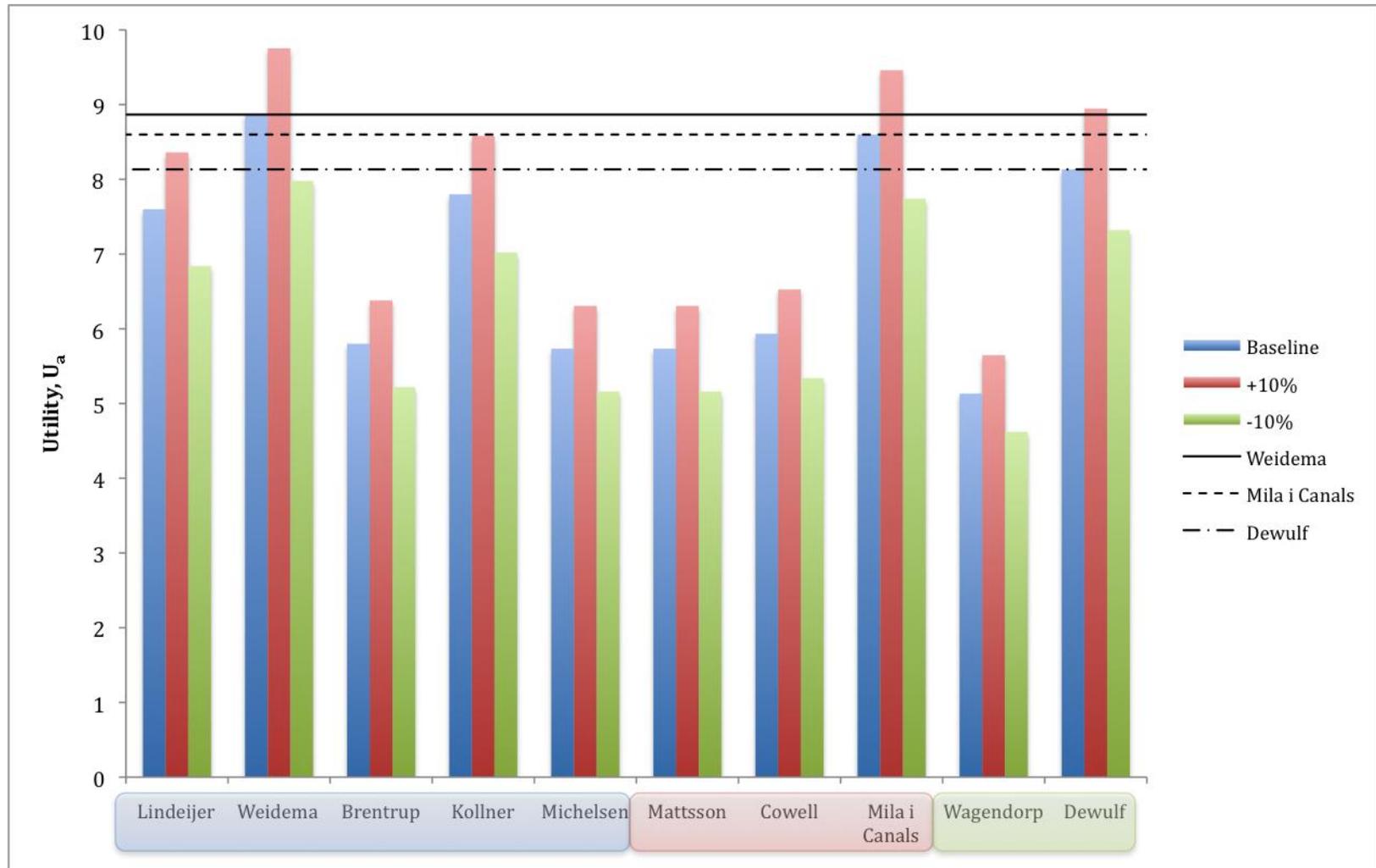


Figure 3.2: Results of overall value and sensitivity to changes in initial criterion scores. The blue, red, and green highlights denote methods based on biomass/biodiversity, soil, and thermodynamics respectively.

Figure 3.1 illustrates the utility score for each method and how each criterion contributed to that score. The first column represents the baseline case, that is, the weights presented in Table 3.3 and the Baseline column of Table 3.5. The other three columns represent three alternate weighting schemes and demonstrate the sensitivity of the method selected to change in weight. The alternate schemes were generated by weighting all criteria equally, and by swapping the weights of two criteria, holding the rest constant. The swapping of weights was done to allow results to be displayed on the same chart easily. The weights (w_i) for each criterion and scheme are listed in Table 3.4. The method of Weidema and Lindeijer (2001) dominated in all cases except for two, Equal and Valuation (not shown). It tied with the method of Mila i Canals et al. (2007b) in the Equal scheme and lost to the method of Mila i Canals et al. (2007b) in the Valuation scheme.

Table 3.5: Alternate weighting schemes for analysis of sensitivity of utility to weights

Criterion	Weighting scheme			
	Baseline	Equal	Descriptive	Objective
1. Descriptive	.267	.200	.333	.267
2. Linear	.133	.200	.133	.133
3. Objective	.200	.200	.200	.333
4. Valuation	.067	.200	.067	.067
5. Practical	.333	.200	.267	.200

Figure 3.2 illustrates the sensitivity of the preferred method to a 10% increase or decrease in utility as a result of scoring inconsistency. As in Figure 3.1, the baseline case used the scores from Table 3.4 and the weights from Table 3.3. Individual method increases of 10% in utility could allow for the methods of Mila i Canals et al. (2007b) and Dewulf et al. (2007) to be preferred over that of Weidema and Lindeijer (2001). A

combination of several increases and decreases could allow five different preferred methods. For application to the case study, the methods of Weidema and Lindeijer (2001) and Mila i Canals et al. (2007b) were selected based on the results of the sensitivity analysis.

5. DISCUSSION

5.1 Results of MCDA

Figure 3.1 demonstrates that the method of Weidema and Lindeijer (2001) has the highest overall value for the initial, descriptive, and objective weighting schemes, and is thus a preferred method to apply to the proposed case study. Under the equal weighting scheme, the method of Mila i Canals et al. (2007b) had an overall utility equal to that of Weidema and Lindeijer (2001). Under a weighting scheme in which the weights for the Valuation and Practical criteria were swapped, the method of Mila i Canals et al. (2007b) was preferred. This demonstrates that the overall utility showed low sensitivity to changes in the weights. The method preferred was sensitive to changes in utility of 10%.

There is some bias in the selection criteria that led to the preferred methods. In particular, different criteria favor different methods partly based on the number of indicators that the method includes. For example, the valuation criterion favors smaller numbers of indicators, as in the valuation step of LCA larger numbers of indicators are more difficult to aggregate (Hertwich et al. 2002). The practical criterion also favors smaller numbers of indicators, because more indicators require more data, and limited data availability is a frequently cited obstacle to land use impact assessment. This forces

the LCA practitioner to to make assumptions, use models, or abandon an impact assessment method altogether.

The descriptive criterion, however, is more likely to favor methods with two or more indicators, as indicators generally apply to a single impact category. While Weidema and Lindeijer (2001) and Mila i Canals et al. (2007b) make arguments to apply a single indicator to two impact categories, a high score in this criterion suggests multiple indicators.

The linear and objective criteria are not intrinsically tied to the number of indicators based on the scoring system. They simply depend on the choices that the land use impact modeler made in constructing the method.

One aspect of the practical criterion that was ignored by this study was the regional nature of the data sets used in many of the proposed methods. The data and thus the models that generated characterization factors to apply to the inventory data are based on country or regional data (Müller-Wenk. 1998, Lindeijer. 2000a, Köllner and Scholz. 2008): Swiss lowlands or Europe, for example. This omission was based on the assumption that comparisons between the locations and eco-regions used in the studies and the locations and eco-regions of interest in this study could easily be drawn based on similarities between eco-regions and land use types. If this assumption had not excluded this aspect of the proposed methods, it is possible that the practical criterion scores would have changed significantly. However, the nature of screening-level LCA could justify this assumption.

Regardless of the level of the LCA, the goal and scope of the study are vital in determining the rest of the process (Mila i Canals et al. 2007a, Hertwich. 2002). Likewise the MCDA utilized in this study is affected by the goal and scope. For this study, a hypothetical LCA is proposed, and the two preferred methods from the MCDA have been applied to it. However, as the goal and scope are essential to the rest of the process, a starting point is necessary (Munier. 2004).

As a result of this starting point, the MCDA criteria and weights are based on the decisions of one person. While some aspects, such as the practicality of a method may be important for all studies, not all criteria selected for this study are applicable to every case study.

While MCDA cannot remove the ultimately qualitative value judgments expressed in the process, it does seek to make them explicit, and ideally the process reflects the opinions and judgments of an expert panel or the parties responsible for the study (Belton and Stewart. 2002). In this case it would be helpful to have the input of appropriate stakeholders to understand the impacts they consider important to analyze the development of the bioenergy sector in SC.

5.2 Application to case study

Switchgrass (*Panicum virgatum*) is a perennial warm-season grass that has shown promise both as a feedstock for cellulosic ethanol and for co-firing with coal in traditional power plants (McLaughlin and Kszos. 2005). Switchgrass also shows potential to sequester significant amounts of carbon in the soils in which it grows (McLaughlin and Kszos. 2005, Tolbert et al. 2002). Some of these properties are due to the perennial nature

of the grass, which requires no tilling once it is established. This stands in contrast to many annual crops that disturb soil significantly in planting and harvesting. Switchgrass has also shown potential to grow with limited fertilizer inputs and in marginal soils (Tolbert et al. 2002, Ma et al. 2000).

The Pee Dee region of SC includes $2.54 * 10^5$ hectares of cropland in production (U.S. Department of Agriculture, National Agricultural Statistics Service. 2009) and $.191 * 10^5$ hectares of land in CRP (Barbarika. 2009), representing the most current data available.

The method of Weidema and Lindeijer (2001) uses indicators for ecosystem quality based on Net Primary Productivity (NPP) and biodiversity. Where the impact of occupying the land is calculated using the equation

$$I_{occ} = A * t_i * \Delta Q / s \quad (2)$$

Where I_{occ} is the occupation impact, A is the area occupied or transformed, t is time, the subscript i can refer to either the activity or relaxation, ΔQ is the change in quality as a result of the activity or relaxation, and s is the slope factor for the activity or relaxation. Since the growth of switchgrass on land already used to grow crops does not require the conversion of land, the impacts of relaxation are not allocated to this activity per Weidema and Lindeijer (2001). However, the conversion of wooded CRP land does require the consideration of relaxation time. Tables 3.5 and 3.6 walk through the calculation of impacts using the method of Weidema and Lindeijer (2001) for cropland and converted CRP land in the Pee Dee region.

Table 3.6: Impact of occupying existing cropland for production of switchgrass in the Pee Dee region.

		Productivity	Biodiversity
Quantity	Variable (unit)	Activity	Activity
1. Area	A (m²)	2.54E+09	2.54E+09
2. Time	t_{act} (y)	1	1
3. Potential Quality	Q_{pot} (gC/m²-y or biodiversity)	700	148
4. Actual Quality	Q_{act} (gC/m²-y or biodiversity)	360	0
5. Slope Factor	s	1	1
6. Quality Reduction	ΔQ (gC/m²-y or biodiversity)	340	148
7. Impact	I_{occ} (gC or biodiv. m²-y)	8.65E+11	3.77E+11

In tables 3.6 and 3.7, the areas came from agricultural statistics (Barbarika. 2009,U.S. Department of Agriculture, National Agricultural Statistics Service. 2009) for the counties in the region: Darlington, Dillon, Florence, Lee, Marion, Marlboro, and Williamsburg. The time of activity was one year, and the relaxation times were interpolated for 34° N latitude from data in Weidema and Lindeijer (2001).

Table 3.7: Impact of occupying CRP land for production of switchgrass in the Pee Dee Region

		Productivity		Biodiversity	
Quantity	Variable (unit)	Activity	Relaxation	Activity	Relaxation
1. Area	A (m²)	1.91E+08	1.91E+08	1.91E+08	1.91E+08
2. Time	t_i (y)	1	118	1	708
3. Potential Quality	Q_{pot} (gC/m²-y or biodiversity)	700	700	148	148
4. Actual Quality	Q_{act} (gC/m²-y or biodiversity)	360	360	0	0
5. Slope Factor	s	1	2	1	2
6. Quality Reduction	ΔQ (gC/m²-y or biodiversity)	340	170	148	74
7. Impact	I_{occ} (gC or biodiv. m²-y)	6.49E+10	3.83E+12	2.83E+10	1.00E+13
8. Totals	(gC or biodiv. m²-y)		3.89E+12		1.01E+13

In tables 3.5 and 3.6 the slope factor of 1 refers to a constant quality during occupation and the slope factor of 2 a changing quality during relaxation per Weidema and Lindeijer (2001). The potential qualities were based on the same data used in Weidema and Lindeijer (2001), treating the Pee Dee region as temperate woodland, consistent with arguments from Weidema and Lindeijer (2001) and maps of the region (Griffith et al. 2002, Olson et al. 2001, Mutke and Barthlott. 2005). This led to a potential productivity quality for the region of 700 gC/m²-y (Weidema and Lindeijer. 2001) and potential biodiversity Quality of 148. Biodiversity quality was calculated using equation (3) adapted from Weidema and Lindeijer (2001) and using data from the same.

$$Q_{pot} = (SR / SR_{min}) * (A_{pot,max} / A_{pot}) * (A_{exi} / A_{pot}) \quad (3)$$

Where *SR* is species richness as determined from global vascular plant species maps, the minimum value being representative of deserts and tundra. *A* is the area of the biome with the subscript *pot*, *max* being the potential area of the largest biome—boreal forest, and the subscript *pot* being the potential area of the biome of study. The subscript *exi* refers to the existing area of the biome of study (Weidema and Lindeijer. 2001).

The actual qualities, line 4 in Tables 3.5 and 3.6, for productivity resulted from average switchgrass yield for the southeastern US (McLaughlin and Kszos. 2005), a biomass to NPP ratio of .45 (Weidema and Lindeijer. 2001), and a carbon content of 50% (Bransby et al. 1998). The actual biodiversity value of 0 reflects the lack of original species left in the study area (Weidema and Lindeijer. 2001). Quality reduction, line 5, is the difference between Q_{pot} and Q_{act} . The impact, line 6, was calculated using equation 2, and the results were in grams of carbon and biodiversity weighted m²-y. The impacts to

productivity are considered a carbon emission for the system, as it represents carbon that would have been absorbed by the reference situation.

The method of Mila i Canals et al. (2007b) is based measurements of the soil organic matter (SOM) for the studied site. If site measurements are not available, model predictions tailored to the site are preferred to estimations based on literature data for similar conditions. The method is based on comparing the studied system to a reference situation that is most likely to exist in the absence of the studied system. The method requires the area occupied, SOM at the start and end of the activity, SOM during the occupation process, SOM for the reference situation, SOM for the relaxation period, and the potential maximum SOM for the site. The quality indicator is then the difference in SOM for the occupation added to the difference in SOM for the relaxation period (Mila i Canals et al. 2007).

Based on the screening-level nature of the study, estimations based on literature were most appropriate. This study assumed that all of the soil in the Pee Dee region was a sandy loam type, and that land currently in production would grow corn if not switchgrass. Sandy loam soils are present in the area (Soil Survey Staff. 2010), and corn is a major crop in the region (U.S. Department of Agriculture, National Agricultural Statistics Service. 2009). The treatment of CRP land was excluded due to data limitations. The relaxation period was also omitted due to data limitations.

The average rate of carbon accumulation in soils under switchgrass for a wide variety of conditions in the southeast US is .78 MgC/ha-y, assuming an established switchgrass stand (McLaughlin and Kszos. 2005). The average soil organic carbon (SOC)

content in sandy loam soils under corn for trials in Alabama is 7.28 MgC/ha. SOM is 1.724 times greater than SOC (Mila i Canals et al. 2007). This led to a starting SOM level for the reference situation (corn) and the switchgrass of 12.6 MgC/ha. The soil under corn was assumed to remain constant for the year while the switchgrass soil accumulated carbon at a rate of .78 MgC/ha-y; which resulted in a difference at the end of the year between the two scenarios of .78 MgC/ha. For the 2.54E+5 ha of land currently in production, this represents a carbon sequestration of 1.98E+5 MgC for the year. Please note the result as calculated in the method yields MgC rather than gC as in the previous method.

Table 3.8: Summary of impacts associated with the production of switchgrass in the Pee Dee Region.

	CRP	Cropland	Total
Area (m ²)	1.91E+08	2.54E+09	2.73E+09
Biodiversity Impact (m ² -y)	1.01E+13	3.77E+11	1.04E+13
Productivity impact (gC)	3.44E+12	7.63E+11	4.20E+12
Carbon Sequestration (gC)	n.a.	1.98E+11	n.a.

Comparing the two methods can reveal the tradeoffs associated with growing switchgrass in a given year, as shown in Table 3.8. While the CRP land is an order of magnitude less in its area than the cropland, its impact on biodiversity is two orders of magnitude greater than that of cropland. For productivity impact, CRP land is one order of magnitude greater than that of cropland. The productivity impact of cropland is on the same order of magnitude as its carbon sequestration. While the impact of the production of switchgrass upon life support functions of soil based on SOM were not calculated due to data limitations, the preparation of that land would have required significant preparation of the land to plant switchgrass. Assuming temperate woodland occupied the

CRP land, significant clearing and subsequent loss of carbon from the soil would need to be included in the SOM balance. Without data, it is impossible to draw a conclusion, but it is possible that the SOM of the woodland soil would be greater than that of switchgrass soil, which would result in an initial loss of SOM followed by recovery.

These results highlight the importance of land transformation in the results of a land use impact method. In the method of Weidema and Lindeijer (2001) land transformation requires the inclusion of relaxation times, which dominate the results over short occupation times. The results also highlight the importance of the reference situation. The method of Weidema and Lindeijer (2001) uses the local biome as the reference situation, while the method of Mila i Canals et al. (2007b) uses the most likely alternative use of the land. This means that the reference situation is the same for both cropland and CRP land within the region under the method of Weidema and Lindeijer (2001), but the method of Mila i Canals et al. (2007b) would have a different reference situation for cropland and CRP land. The most likely alternative use for CRP land other than switchgrass is still CRP. This makes the direct comparison of CRP and cropland difficult.

Similarly the nature of the impacts associated with land use makes comparison of results even with a method difficult. The two metrics in the method of Weidema and Lindeijer (2001) have very different units: gC and biodiversity weighted m^2 -y. These different units would require a weighting step similar to MCDA to assign importance to the results.

The data that go into the two methods add more complexity to interpreting results. There is uncertainty associated with all of the data used in these methods, as noted by the authors themselves in their respective papers (Mila i Canals et al. 2007b ,Weidema and Lindeijer. 2001). Additionally, the data used in calculating the SOM for the switchgrass production do not match entirely with the tabulated data in Mila i Canals et al. (2007b). The maximum sequestration rate listed for energy crops grown on arable land was .6 MgC/ha-y (Mila i Canals et al. 2007) while the rate reported for the southeast US was .78 MgC/ha-y (McLaughlin and Kszos. 2005). This is not attributable to the difference in definition between SOC and SOM.

The production of switchgrass is a natural system, and many factors like initial soil quality, soil type, nitrogen application, previous land use, rainfall, irrigation, cutting schedule, and tillage can influence switchgrass yields, nutrient partitioning, and carbon sequestration (McLaughlin and Kszos. 2005, Bransby et al. 1998). These factors would influence the many simplifying assumptions that were necessary to conduct the screening-level analysis as proposed.

Even after conduction MCDA to select two methods to apply to this case study, limited data availability hindered the full calculation of impacts using the method of Mila i Canals et al. (2007b). This may suggest that using MCDA in LCA should overlap significantly with the LCI phase in order to better assign scores to impact assessment methods.

CONCLUSIONS AND RECOMMENDATIONS

LCA is an evolving tool that is currently struggling to deal with the incorporation of land use impacts into the LCA structure. These impacts are significant and should be included in LCA (Lindeijer et al. 2002b, Blonk et al. 1997, Müller-Wenk. 1998, Mila i Canals et al. 2007a) especially for cases that involve the production of energy products from bio-based sources. This type of production can have significant tradeoffs (Miller et al. 2007) and comparison to petroleum energy products would benefit from the consideration of the entire life cycles of both types of production.

Data availability continues to be a stumbling block for incorporating impacts of land use into LCA; several times in the methods reviewed, the authors were unable to complete their case studies with the indicators as proposed due to data availability issues (Michelsen. 2008, Mattsson et al. 2000, Cowell and Clift. 2000). The practitioner is then left to deal not only with the availability of data for their own studies, but also the applicability of the method. The practitioner must decide at what point a method is missing too many of its indicators to be of use, or whether simply to exclude certain indicators from use in the study.

The practitioner must also manage conflicting recommendations. In an editorial response to the debate that followed the release of the UNEP-SETAC Life Cycle Initiative framework for land use in LCA (Mila i Canals et al. 2007a), Baitz (2007) suggests that active practice should play an important part in the development of land use impact assessment methods for LCA. Conversely, in a guide for LCA of conservation

agriculture, Guinée et al. (2006) recommend that that land use impacts be restricted to the inventory result of the product of area and time—data that are readily available.

It is possible that the use of MCDA to identify potential land use impact assessment methods could play a key role in bringing the practice of assessing land use impacts firmly into method development. By identifying methods in a preliminary survey of available data and other preference criteria, a few promising alternatives could be identified for further investigation without requiring the time or resources of a full LCA. The alternatives could then be further examined for use in a subsequent full LCA.

As future work, the repetition of the case study with more region-specific data could offer valuable information to the appropriate stakeholders as to the impacts of pursuing bioenergy production based on switchgrass, but it would require substantial assumptions of how distribution infrastructure may evolve in order to complete a full LCA. Additionally the future work of this study includes encouraging discussion and debate in anticipation of the guidelines from the UNEP-SETAC Life Cycle Initiative regarding the inclusion of land use impacts in LCA, the end of whose scheduled working period is May 2010.

CHAPTER 4

CONCLUSIONS

The first two objectives of this study: comparing and contrasting land use impact assessment methods in LCA and applying said methods to a representative case study were accomplished. The final objective of encouraging discussion in the larger LCA community remains. This study populated a list with possible methods to assess the impacts of land use in LCA, and by applying MCDA to that list, two methods emerged as candidates for application to the case study of switchgrass production in the Pee Dee region of SC as a bioenergy feedstock. One method uses two indicators: one based on biomass production and one based on three biodiversity factors. The second method, less favored in the MCDA, uses a single indicator based on soil organic matter. These methods together incorporated the three impact categories deemed important for the study: impacts to biodiversity, biomass production, and soil quality. The final objective is contingent on making this work known. While the larger problem of incorporating land use into LCA has not been solved, this study does represent a contribution to the field.

Both the literature review and the MCDA are potentially important to the advancement of land use in LCA. The community actively involved in developing and refining land use impact methods is relatively small: the same names appear on many different method proposals. As such, the wider community of LCA practitioners can benefit from a critical review of the body of literature, simply as a reference to potential methods by which they can include land use in their studies. This is still an emerging field.

Likewise, the MCDA offers a method to select one of those methods based on criteria that can be tailored to each individual case study. MCDA can be used to incorporate the value judgments of a panel of legislators, for example, if the study is undertaken to support public policy decisions. MCDA is also amenable to the iterative nature of LCA, as the results of the MCDA can be used to shape the goal and scope of both an LCA or another round of MCDA.

The future work of this study includes publishing both Chapter 2 and Chapter 3 as individual papers in peer-reviewed journals, as both represent contributions to the field. By publishing this work as two papers, and by presenting this material at conferences the final objective laid out in Chapter 1 may be achieved.

APPENDICES

Appendix A

Justification for criterion scores for land use impact assessment

In Lindeijer et al. (2000a), the authors proposed a method using two indicators: one based on numbers of vascular plant species, and one based on amount of biomass available to the environment after human harvest, free Net Primary Production (fNPP). This method addressed two impact categories, and both are linear and quantitative. Four scores resulted from this method, but were not aggregated to a single score (Lindeijer. 2000a). Data on fNPP were not available; vascular plant species data were available from Mutke and Barthlott (2005) and Kier et al. (2009).

In Weidema and Lindeijer (2001), the authors proposed two main indicators Net Primary Production, and a combination of three biodiversity factors. This method also addressed two impact categories, and the indicators were linear and quantitative. Two scores resulted after impact assessment. Data were available from the publication and cited sources Weidema and Lindeijer (2001).

Brentrup et al. (2002) proposed a method based on a qualitative description of the naturalness of the studied area as a proxy for biodiversity and natural resources. The method used a linear model, and it included a weighting step. To apply the method required land use type, area and time occupied, and the location of the studied area (Brentrup et al. 2002), but some key factors only were calculated for Europe.

Köllner and Scholz (2007) and (2008) were a two-paper method proposal based on vascular plant species and also includes data on threatened plant species, mosses, and mollusks, which addresses the biodiversity impact category. The relationship between the inventory and impacts is linear and quantitative, and adequate data exists to use the method; however, some important factors were only calculated for Europe. The authors noted that for policy-level decisions, larger-scale regions needed consideration (Köllner and Scholz. 2007, Köllner and Scholz. 2008).

Michelsen (2008) based the proposed method on Weidema and Lindeijer (2001), but modified the method: the biomass indicator was dropped, and the biodiversity factors were altered. Michelsen rejected vascular plant species as a biodiversity indicator, and the method implemented differed from the method proposed due to data availability. Two of the three factors were qualitative, and one was very specific to the area studied. The method applied to one impact category (Michelsen. 2008). Data for two of the factors was available (Griffith et al. 2002, Olson et al. 2001).

Mattsson et al. (2000) proposed seven soil quality indicators, one biodiversity indicator, and one qualitative landscape value indicator. Two of the seven soil indicators were linear, and several of the indicators were not used in case studies due to data availability. Soil quality can be argued to apply to both soil quality and biological productivity (Mila i Canals et al. 2007b, Cowell and Clift. 2000), and coupled with the biodiversity and landscape value indicators, the method applies to more than the three identified impact categories. The large number of indicators make aggregation difficult (Mattsson et al. 2000). Soil data for half of the indicators were available in Geographical

Information System (GIS) form but would require significant aggregation (Soil Survey Staff.).

Cowell and Clift (2000) proposed a method based on a variety of soil quality indicators, but ultimately reduced them to three based on data availability. The justification for weights of this method and the data used to apply it are essentially the same as for Mattson et al. (2000).

Mila i Canals et al (2007b) proposed a method based on the single indicator of soil organic matter (SOM). This was quantitatively and linearly tied to soil quality and biomass production of interest in this study. The authors linked the indicator to Life Support Functions, the variety of functions that land supplies in the ecosystem. The single indicator allows for easy valuation in the scheme of a larger LCA (Mila i Canals et al. 2007b). Data were available from sources in the paper (Mila i Canals et al. 2007b) and GIS data (Soil Survey Staff. 2010).

Wagendorp et al. (2006) proposed a method based on ecosystem thermodynamics. This approach applied to the biomass production, as the possible indicators apply to the conversion and degradation of energy by the ecosystem; however, the method is not fully developed and requires extensive remote sensing data (Wagendorp et al. 2006).

Dewulf et al. (2007) proposed a method based on ecosystem thermodynamics: useful energy denied the lowest trophic level. This method was, therefore, linked to biomass productivity, although indirectly. The model was linear, quantitative, aggregated to a single indicator, and used available data: area, time, and land use type (Dewulf et al. 2007).

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