MODELING THE NET GREENHOUSE GAS BALANCE OF PROJECTS THAT DISPLACE GASOLINE WITH WOOD ETHANOL FROM SHORT ROTATION TREE PLANTATIONS

by

CATALIN RISTEA

Dipl. Ing. Wood Industry, Universitatea Transilvania, 1993
M.Sc. Forestry, The University of British Columbia, 2001

A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF

DOCTOR OF PHILOSOPHY

in

THE FACULTY OF GRADUATE AND POSTDOCTORAL STUDIES

(Forestry)

THE UNIVERSITY OF BRITISH COLUMBIA
(Vancouver)

April 2014

Abstract

Projects that establish fast growing tree plantations and substitute gasoline with ethanol from the resulting wood biomass have the potential to reduce atmospheric carbon dioxide and other greenhouse gas (GHG) emissions and to increase terrestrial carbon (C) stocks. However, current methodologies for evaluating the net GHG balance of biofuel projects do not consider specifically the life cycle impacts of biogenic C dynamics and the emissions from decomposition of dead organic matter (DOM).

This dissertation proposes the Carbon Balance and Biomass to Biofuel Optimization planning model (C3BO), which determines the net GHG balance of biofuel projects on a life cycle basis from initial land-use change, establishment of plantations and construction of biorefinery, through conversion of biomass into ethanol and final use in internal combustion engines. The novel approach of the C3BO model is the inclusion of initial C stocks, soil organic matter and biogenic C dynamics to the project GHG balance calculations.

C3BO model results show that the GHG balance of biofuel projects is most sensitive to initial carbon stocks and biorefinery emissions. The potential direct land-use change impacts on initial C stocks (including the initial biomass removal and the affect of the project on the emissions from decay of soil organic matter) can affect the life-cycle net GHG balance, clearly indicating that they need to be included in life cycle analyses of biofuel projects.

The magnitude of the reductions in emissions, compared with the fossil fuel baseline, is highly dependent on the length of the project time horizon: the GHG balance changes with project length, and the impact of input variables also changes with time.

Biofuel projects can produce fewer emissions but they can also result in more emissions than the displaced fossil fuel system. The viability of biofuel projects as worthwhile climate mitigation strategies depends on project-specific conditions that need to be properly assessed on a project-by-project basis.

This study also suggests that ethanol production cost is twice as sensitive to conversion efficiency than to biomass yield. Improving conversion efficiency will result in much larger benefits than improving biomass yield, in terms of reducing ethanol production costs.

Preface

This dissertation is original, unpublished work by the author, Catalin Ristea, except as noted below.

The research questions and objectives, the scope of the dissertation, the various research methodologies and analysis approaches, and the interpretation and context of research results, have been discussed with, and much improved based on the feedback of, my Ph.D. Supervisory Committee members: Dr. Thomas C. Maness, Dr. Gary Q. Bull, Dr. Shawn D. Mansfield, and Dr. John D. Nelson. A series of manuscripts based on this dissertation are being prepared for subsequent publication in peer-reviewed journals, to be co-authored with the Supervisory Committee members based on their respective intellectual contribution to the research chapters.

Jake Eaton and Dr. Brian Stanton from GreenWood Resources Inc. have contributed to the research work of CHAPTER 2, in particular with the discussion of parameters for the various poplar biomass production strategies, plantation establishment and management activities, with technical and engineering data from their industrial poplar plantations, and with important feedback on chapter manuscript draft.

Table of Contents

List of Tables

List of Figures

Acknowledgements

First and foremost I want to thank my advisor, Professor and Dean Thomas C. Maness. He has taught me how good research is done and how to prioritize for the important things that matter. His support throughout my graduate life has put me on a journey that changed my life for the better. I appreciate all his contributions of ideas and funding to make my Ph.D. experience stimulating and productive. His pragmatic no-nonsense approach to the research process was inspirational and motivational for me, even during tough times during the Ph.D. program. I am also thankful for the example he has provided as a successful professor and leader of academic and research groups, and a builder of things in general.

The members of my Ph.D. Supervisory Committee, Dr. Gary Bull, Dr. Shawn Mansfield, and Dr. John Nelson, have contributed immensely to my development as a researcher and thinker and to my professional time at the University of British Columbia.

I want to thank Professor Gary Q. Bull who continuously tried to prepare me for defending my ideas, always offering a different perspective and a suggestion for something that I might have missed or I should be aware of; moments at The Old Barn and TW&H will be remembered for a long time.

The good advice, support and encouragement of Professor Shawn D. Mansfield have been invaluable on both an academic and personal level. He kept a sense of humour when I lost mine, and showed me that good things come to those who work hard and keep a high standard of their academic pursuits. I am extremely grateful for his contribution to my funding opportunities, as well.

I would like to thank Professor John D. Nelson for his forthcoming, direct, and uncompromising support and feedback. I appreciate his guidance and helpful suggestions, and for constantly showing me that good research does not have to be complicated, and good discussions need not be long.

Dr. Thomas Sullivan deserves thanks for nurturing my love of wisdom and for mentioning a few books on scientific thought that have changed my world outlook. Yes, of course, for our conversations over coffee and cookies on empirical falsification as a way of pursuing knowledge as well!

My work for this dissertation has been supported in part by the Pacific Institute for Climate Solutions through a Graduate Fellowship. This support is gratefully acknowledged.

Special thanks to Jake Eaton (you left this world way too soon…) and Dr. Brian Stanton at GreenWood Resources Inc. for their intellectual and material support, helpful suggestions, and for sharing their practical knowledge, experience, and industrial data on short rotation poplar production strategies.

Many thanks to Dan Carson when working for Kruger Inc., to Michael Carlson when working for the BC Ministry of Forests, and to Bill Berguson at the Natural Resources Research Institute, U. of Minnesota, for sharing their knowledge on fast growing poplars and sharing a little of their busy time with me.

Without the magic powers of the python and java wizard Dr. Ovidiu Toader, the computer implementation of the optimization model would not look the same; he also helped me immensely build confidence and strengthen my argument by constantly challenging my assumptions; many thanks, Oviii. I was also very lucky to have crossed paths along my journey with Dr. Olaf Schwab, Dr. Cristian Palma, Justin Bull, Dr. Christina Staudhammer, Dr. Darrell Wong, Francisco Vergara, Heather Akai, Dr. Marian Marinescu, Dr. Mihai Pavel, Dr. Ciprian Lazarescu, Cosmin Man, Dr. Mona Berciu, Dr. Luiz Oliveira, Dr. Bruce Lehmann; their sayings, doings, and kind spirits have touched me in good ways.

I wish to thank my wife, Anca, whose love, encouragement, nourishment, energy, support, patience – did I say love? – allowed me to go through what often seemed like a quagmire and finish this journey. She already has my heart so I will just give her a heartfelt "thanks." To my son, Alex, who is bringing so much happiness into my life, for all the lessons that he taught me while I thought I was raising HIM up. To Mirela, Ruxandra, and Bodgan Ristea, Neculae Avram and Anica, Nicoleta Avram, Raluca and Catrinel Gherman, to the Porime family (Nicoleta, Diana, Catalina, Flavia, and Flavius), to the Paduraru family (Peggy, Andrei, Nicholas, Jasmine, and Dragos), to the Puscasu family (Ioana, Anca, Radu, and Adrian), and to all my extended family who have constantly encouraged me. Furthermore, I also want to thank all friends who put up with me through the whole Ph.D. process and helped me with personal challenges, in particular Irina Goga who always had time to simply listen, Doina and Marian Balea, Lucian Tancou, Elena and Gelu Halmaghi, Carmen

Costea, Elena and Catalin Ursu, with whom I shared a dance, a laugh, a movie or two, a rainbow trout or seven. Thank you to all for sharing a little of your life with me. We shall continue to remember that the answer is not always forty two. Please ask the questions.

Finally, I would like to thank mom and dad, who have brought me up into this world, for their unconditional love and support throughout life.

Vancouver, B.C., February 2014 *Catalin Ristea*

Dedication

To Anca, Alex, Silvia, Gheorghe, and Irina, who have brought so much joy to my life.

CHAPTER 1 Introduction

1.1 Background and scope of research

Emissions of greenhouse gases (GHG) from the production and consumption of fossil fuels represent over 56% of anthropogenic GHG emissions, which are very likely contributing to recent global warming trends (IPCC 2007). An essential strategy for mitigating climate change would be to reduce the consumption of fossil fuels, but it is unrealistic to expect liquid transportation fuel consumption to decrease over the next few decades (IEA 2010). In 2004, transport energy use amounted to 26% of total world energy use and the transport sector was responsible for about 23% of world energy-related GHG emissions. Transport energy use in 2030 is forecasted to be about 80% higher than in 2002, and almost all of this new consumption is expected to be in petroleum fuels (IPCC 2007). The sectors propelling worldwide transport energy growth are primarily light-duty vehicles, freight trucks and air travel. The Mobility 2030 study (WBCSD 2004) projects that these three sectors will be responsible for 38, 27 and 23%, respectively, of the total 100 EJ growth in transport energy that it foresees in the 2000–2050 period. As a result, CO_2 emissions will essentially grow in lockstep with energy consumption (IPCC 2007). It becomes therefore important to develop sustainable alternatives to fossil transportation fuels.

Biomass is seen as the key large-scale renewable energy source that can produce competitively priced transportation fuels (Maniatis *et al.* 2012). After the 1970s oil crisis, large research programmes and policies have been established in many countries to encourage the development of non-fossil energy sources, with the key objective of improving energy security. The International Energy Agency, established in 1974, has supported research efforts in biomass productivity as well as technologies for conversion to biofuels. The US Energy Tax Act of 1978 created ethanol tax credits in an effort to decrease the nation's vulnerability to oil shortages and handle how the price of corn had been depressed by agricultural subsidies.

More recently, the US Energy Policy Act of 2005 provided tax incentives and loan guarantees for energy production of various types. This included ethanol from cellulosic biomass, which has the potential to contribute to meeting the demand for liquid fuels

(Rubin 2008)and to contribute to energy and environmental goals (Farrell et al. 2006). The US Energy Independence and Security Act of 2007 and the Renewable Fuels Standard program required renewable fuel to be blended into transportation fuel in increasing amounts each year, escalating to 36 billion gallons by 2022. This mandated use resulted in a significant demand and market for cellulosic biofuels.

However, the Renewable Fuels Standard also required that cellulosic biofuels must have life cycle greenhouse gas emissions at least 60% lower than the baseline petroleum fuel. A vigorous debate ensued in the scientific community regarding the impacts of producing biomass and biofuels (Fargione et al. 2008, Searchinger et al. 2008, Tilman et al. 2009).

The substitution of fossil transportation fuels with renewable biofuels, such as ethanol produced through biochemical conversion from wood biomass grown in short rotation energy crops, was reported as a potentially viable alternative (National Research Council 2009).

The argument in favour of this tactic is that while both biomass-derived biofuels and fossil fuels emit carbon dioxide (CO_2) when used as energy sources, biomass captures and sequesters CO_2 from the atmosphere when it is grown on a sustained cycle. This approach has the potential to both reduce overall atmospheric CO_2 concentrations and increase terrestrial carbon (C) stocks by sequestering C in live biomass and soil. Dedicated energy plantations of fast growing native trees on marginal agricultural lands or unproductive forestlands have the potential to provide a key source of wood biomass for biofuel initiatives.

This creates a unique opportunity to identify the optimal biomass production strategies from afforestation projects that can deliver economic and climate mitigation benefits on a sustained basis, and to evaluate methods for a viable conversion to biofuel products. However, besides the displacement of emissions from transportation fossil fuel production systems, there are other important impacts of these large-scale biofuel systems that need to be considered. Lands that are to be afforested have an existing stock of organic matter and a biophysical GHG flux dynamic, which will be affected by the establishment of fast growing tree plantations; this has the potential to either increase or decrease the total GHG balance of the land base over the project time horizon. The GHG emissions resulting from the production activities of biofuel systems also need to be accounted for on a life cycle basis.

As a result, policy makers are under pressure from stakeholders to devise sound policies for the appropriate management of resources that would create economic opportunities, while reducing greenhouse gas emissions and addressing land-use concerns. However, the effectiveness of these large biofuel production systems – from biomass production to the final use of the biofuel product – is not well understood at the project level, especially from the viewpoint of the impact on the above- and below-ground organic matter stocks throughout the project time horizon. Moreover, biofuel projects could be seen as an opportunity to generate lucrative carbon offsets if the projects are thought to be increasing carbon stocks on the land base and/or reducing the overall GHG emissions to the atmosphere over their lifetime, but there are no generally agreed-upon standards and methods on how to properly account for all the carbon-equivalent uptakes and releases on a full project life cycle basis.

Current methodologies for incorporating specific biofuel issues (*i.e.* impacts on the land base, dynamics of organic matter stocks, greenhouse gas fluxes, temporal scales, project costs and benefits) in systems analysis frameworks – such as life cycle assessment (LCA) and carbon accounting approaches – do not consider these elements in an appropriate spatial and time-dependent manner, and from a full life cycle perspective. Current methods do not take into account all the emissions and sequestration that would not have occurred in the absence of the biofuel project. For example, current methods do not account for the full amount of carbon dioxide that is removed from the atmosphere by a tree plantation, but only for that portion which ends up in the harvested biomass converted to biofuel. Likewise, current methods do not account specifically for the emissions from dead organic matter through natural decomposition. It is not unreasonable to suspect that these limitations therefore are unable to properly quantify the potential of biofuel for climate mitigation, and miss the overarching goal, which is the reduction of overall atmospheric GHG/carbon dioxide stocks and/or increasing the terrestrial organic matter/carbon stocks.

This research dissertation aims to address the knowledge gap with respect to modeling large-scale afforestation for biofuel production systems – encompassing the full cycle from land-use change and biomass production to the final use of the biofuel product – by including as much as possible the emissions and sequestration that would not have occurred in the absence of the project: all the carbon dioxide removals from the atmosphere, all the greenhouse gas emissions from decomposition of organic matter and all other production activities, all the sequestered carbon at the end of the project time horizon, in a time-dependent framework.

The following subsections of CHAPTER 1 contain a literature review of current issues in modeling large-scale production systems of wood biomass to biofuel, the problem definition, the research questions and objectives, and a description of the structure of the dissertation.

1.2 Literature review

The Intergovernmental Panel on Climate Change reported that emissions of greenhouse gases from production and consumption of fossil fuels are a major factor influencing the global climate (IPCC 2007). A proposed strategy for mitigating climate change is to reduce consumption of fossil fuel resources. One way to accomplish this is to substitute fossil fuels with renewable biofuels from biomass, such as wood. The argument is that biofuel use reduces overall atmospheric CO_2 concentrations compared to fossil fuel use, because biomass absorbs significant quantities of CO_2 from the atmosphere during growth, while fossil fuels only emit CO_2 when used for energy.

However, the peer-reviewed literature on biofuel systems modeling suggests that there is a lack of consensus concerning the accounting methods for carbon (and stored organic matter) and GHG emissions fluxes over time, the overall net affect of biofuel production systems on atmospheric GHG emissions, land base impacts, and land-use change effects (Campbell et al. 2009, Hill et al. 2009, Liska and Perrin 2009, Mathews and Tan 2009, Dale 2008, Fargione et al. 2008, Robertson et al. 2008, Scharlemann and Laurance 2008, Searchinger et al. 2008, Dale 2007, Hill 2007, von Blottnitz and Curran 2007, National Research Council 2009).

A thorough review of the published peer-reviewed literature suggests that currently there is no available published study on large scale production systems of dedicated fast growing tree plantations supplying wood biomass for conversion into ethanol that adequately investigated the net carbon and greenhouse gas balances including in live and dead organic matter.

There are, however, studies that investigated either only some aspects of biofuel production, or at different spatial and temporal scales, mainly national and global analyses using generic and aggregated data:

- Various economic models (McKenney *et al.* 2004, van Kooten 2000, Audsley and Annetts 2003, Hamelinck *et al.* 2005, Hendrickson *et al.* 1998, Hendrickson *et al.* 2006, Piccolo and Bezzo 2009, Ramlal *et al.* 2009, Thorsell *et al.* 2004, Phillips 2007) including computable general equilibrium models for national or global economies using the general equilibrium paradigm for economic and modeling (Melillo *et al.* 2009),
- Cost-benefit analyses (van Kooten *et al.* 1999),
- Life cycle analyses of various biofuels (Perez-Garcia et al. 2005, Zah et al. 2007, Young 2003, Spatari et al. 2005, TIAX LLC 2007, Rafaschieri et al. 1999, Liska et al. 2009, Lippke et al. 2004, Macdonald et al. 1997, Mann and Spath 2001, RW.ERROR - Unable to find reference:872, Heller et al. 2004, Heller et al. 2003, Hendrickson et al. 1998, Hendrickson et al. 2006, ICF International 2009, Grant et al. 2008, Field et al. 2001, Fu et al. 2003, Delucchi 2003, Rabl et al. 2007),
- Carbon stocks and modeling (Kull *et al.* 2007, Marshall 2009, Matthews *et al.* 2008, Matthews 2001, Schlamadinger and Marland 1996, Smith and Heath 2006, Smith *et al.* 2007, Tonn and Marland 2007), and,
- Energy input-output studies (Schmer *et al.* 2008, Shapouri *et al.* 2002, Sheehan *et al.* 2004, Chambers *et al.* 1979, Dale 2008, Dale 2007, Farrell *et al.* 2006, Field *et al.* 2008, Gielen *et al.* 2001, Groode and Heywood 2006, Hammerschlag 2006, Matthews 2001, Pimentel 2003, Ptasinski *et al.* 2007).

To describe the background of this research and the context of the published literature on the subject, the following key topics will be discussed in this section:

- Demand outlook for biofuel and current policy context in British Columbia

- Current approaches for modeling biofuel systems on a life cycle basis, quantifying energy inputs-outputs and greenhouse gas balances
- Production of wood biomass from establishment of fast-growing poplar plantations
- Land base impacts of biomass production
- Conversion technologies and biofuel products from wood feedstocks
- Accounting methods for greenhouse gas emissions and carbon

1.2.1 Demand outlook for biofuel and current policy context in British Columbia

The Province of British Columbia has implemented a series of policy and legislation initiatives, including the BC Bioenergy Strategy, the BC Climate Action Plan, and the BC Energy Plan. The November 2007 Greenhouse Gas Reduction Targets Act entrenched the following commitments in law: by 2020, BC will reduce its greenhouse gas emissions by 33% compared to 2007 levels; by 2050, GHG emissions in BC will be reduced by at least 80 per cent below 2007 levels. The province has also instituted in 2008 a revenue-neutral carbon tax. BC aims for provincial biofuel production that will meet 50% or more of the province's renewable fuel requirements by 2020. The BC renewable biofuels requirements policy requires gasoline and diesel fuel sold in British Columbia to have 5% renewable content by volume by 2010 (Climate Action Secretariat, Gov't. of B.C. 2008, BC MoEMPR 2008, BC MoEMPR 2008).

According to a GLOBE Foundation (2007) report, the BC domestic transportation (which represents 87% of gasoline and diesel fuel sales in BC) energy consumption is forecasted to increase to 257 PJ (petajoules) by year 2025. This means that, at a 10% ethanol blend in gasoline, there is the potential for 25 PJ to be derived from biofuels by 2025 in BC, which translates in a provincial demand for 850 million litres/yr by year 2025. BC is reported to have the potential to produce 50 PJ of feedstock energy per year from fast growing trees as energy crops, which could in turn produce 1.25 million litres of liquid fuels; however, no details are given on these assumptions. Two reports by BIOCAP Canada (2006, 2008) suggest that about 4 million dry tonnes of wood can be made available from energy crops in British Columbia, but no details are given on how and when this may be accomplished.

At the national level, the Canadian federal government expressed intent to proceed with regulations requiring a minimum 5% ethanol blend (E5) in gasoline. This mandate would represent about 2 billion litres annually, compared with the 2010 production capacity of the Canadian ethanol plants in operation of 1.7 billion litres (Canadian Renewable Fuels Association 2010). This represents a substantial potential demand for biofuels for transportation such as bioethanol.

It is apparent that there will be in the near future a very significant demand for biofuels produced in BC, especially cellulosic biofuels. However, at this time there are no large-scale fast growing tree energy crops, no commercial-scale cellulosic biofuel biorefineries operating anywhere in the world, and many conversion technologies are still in the early development stages (IEA 2010).

1.2.2 Current approaches for modeling biofuel systems on a life cycle basis, quantifying energy inputs-outputs and greenhouse gas balances

Published methodologies of greenhouse gas balance for biofuel projects on a life cycle basis use an energy input-output approach, and infer from the amount of energy consumed by various activities the resulting equivalent greenhouse gas emissions. The Argonne National Laboratory's Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) model (Wang *et al.* 2007) is an energy input-output balance model; it first calculates consumption of total energy and inputs, and based on the energy results, it then calculates the emissions of CO_2-equivalent GHG gasses. GREET calculates total greenhouse gas emissions (gCO_2e/MJ) on a CO_2 equivalent basis per unit of energy (MJ) for a given fuel. The functional units are energy consumption per mile, and emissions per mile. It considers the fuel cycle on a "well-to-wheels" (WTW) basis, separately for the well-to-tank pathway that includes all steps from feedstock production to final finished product, and for the tank-to-wheels pathway that includes actual combustion of fuel in a motor vehicle for motive power. The California Environmental Protection Agency – Air Resources Board has somewhat modified the GREET model in order to calculate the energy use and greenhouse gas (GHG) emissions associated with a WTW analysis of ethanol produced from farmed trees by fermentation (California Environmental Protection Agency 2009). They include a preliminary value for emissions from direct land-use change (2.40 gCO_2e/MJ), but no details of the calculations are given.

Other energy input-output models include: the Lifecycle Emissions Model (LEM) developed by Delucchi (2003) and its version GHGenius adapted for Canadian conditions by $(S\&T)^2$ Consultants (2003, 2011); the Biofuel Energy Systems Simulator (BESS) model for corn ethanol (Liska *et al.* 2009); Ecobilan (Ecobalance) Price Waterhouse Coopers' commercial life-cycle modeling tool, TEAM, and its companion lifecycle database, DEAM; the Energy and Resources Group Biofuel Analysis Meta-Model (EBAMM) developed by Farrell *et al.* (2006); SimaPro by PRé Consultants (2012).

While the energy input-output models can describe well the energy embedded into biomass, fossil fuels, and other raw input materials, and the energy consumed by the various project activities, these models are by design not necessarily well equipped to consider important aspects of biogenic carbon dynamics, such as sequestration of carbon from the atmosphere by growing biomass, transfer of biogenic carbon between live and dead organic matter pools, and emissions of greenhouse gases through natural processes of decomposition of organic matter.

Another limitation of these energy input-output models, which is also typical of Life Cycle Assessments (LCA) methodologies in general, is that they do not provide a time-dependent analysis of biofuel systems, but instead calculate the energy inputs and outputs and greenhouse gas impacts as a snapshot in time, an average of the values across the project time horizon (in fact many of the models do not consider the notion of time horizon).

1.2.3 Production of wood biomass from establishment of fast-growing poplar plantations

Key potential sources of wood biomass feedstock production have been identified for British Columbia (BIOCAP Canada 2006, BC MoEMPR 2008, BIOCAP Canada 2008) from marginal agricultural lands and underutilized/unproductive forest lands: dedicated energy crops from short rotation plantations (such as fast growing hybrid poplars). One of BC's major resource strengths is its forest and less productive agricultural land base, and the enormous potential for production of renewable woody biomass from this land base offers the province unique opportunities for development of both optimized lignocellulosic feedstocks and the technologies needed to convert these feedstocks into biofuel such as ethanol (GENOME BC 2007).

British Columbia has a huge genetic diversity in its native *Populus* species and small plantations of both native and hybrid trees have also been established in the province. Marginal agricultural or range lands – land that was previously used for agriculture or pasture but that has been abandoned or underutilized and not converted to forest or urban areas – might offer a significant potential for yielding biomass energy that reduces carbon dioxide in atmosphere and avoids competition with food production (Field *et al.* 2008). Poplar plantations require relatively little energy input (unless required for irrigation), can be very high yielding, have short rotation times, and are flexible in terms of where they can be deployed. In British Columbia, marginal agricultural lands that could be considered as suitable for afforestation are those associated with forage production and pasture. In BC's Agricultural Land Commission classification system, these lands would fall under classes 4 and 5, although suitability will depend on the value of lands in their current agricultural activity. In British Columbia, 1.41 million hectares are for pasture or grazing within farm holdings. In addition, an estimated 10 million hectares (8.5 million of which are Crown Land) are classified as open or forested grazing land used by the ranching industry (BC Stats 1999). Afforestation projects by their nature involve a conversion from one land use to another. Typically, in British Columbia, that requires a change in tenure. Governments have the option to subsidize such forest plantations, however, important issues have been identified and need to be addressed in such options (Bull *et al.* 2006).

Dedicated energy crops from fast growing trees are a long-term feedstock option for marginal agricultural or range lands. Cellulosic crops can be grown as more complex species mixes, including native polycultures (Tilman *et al.* 2006) grown for additional conservation benefits. The trade-offs between production needs, carbon sink enhancement and biodiversity are facilitated through the use of inherent genetic diversity in planting stock, and creating a patchwork of crops and ages to create structural diversity (Robertson *et al.* 2008).

In Canada, the woody species used or considered most often for purpose-grown biomass are the poplars (Populus spp.) and willows (Salix spp.). Poplars are native to British Columbia, have inherently fast growth rates and wood that is easier to convert to fermentable sugars than conifers using current bioprocessing technologies (Douglas and Mansfield 2009). Dedicated energy crops with enhanced genomic or transgenic characteristics would represent a significant step forward, as to improve the pre-treatment to destabilize the lignin and hemicellulose, and make cellulose accessible to cellulase enzymes (Coleman *et al.* 2008), understanding cellulose biosynthesis (Joshi and Mansfield 2007), or affect the carbon allocation in hybrid poplar (Coleman *et al.* 2007).

Much of the research regarding poplar for biomass production has concentrated on short rotation intensive culture crops with multiple end products in mind. Most high density poplars planted in Canada to produce pulpwood fibre, rather than 100% biomass feedstock, and the biomass fibre is the by-product. The Province of British Columbia recognizes intensively-managed Populus plantations, which are short rotation intensive crops, as primary agricultural production, but imposes a maximum rotation length of 12 years (BC Assessment 2007, BC Assessment). Although the regulations are beneficial to poplar and willow planting, the restriction of the rotation to 12 years has now proven problematic for poplar plantations. Yield plots in southwestern British Columbia (Carson 2009) show that SRIC hybrid poplar crops planted at 1,100 or fewer stems per hectare do not culminate mean annual increment (MAI) within the 12-year period. A major advantage of classifying *Populus* as primary agricultural production is the flexibility of managing the crop without the regulations that apply to a more traditional forest crop. As a farming operation, there is also the added protection through the Farm Practices Protection (Right to Farm) Act in British Columbia (Queen's Printer 1996), which protection does not apply to forests in the Managed Forestland class (van Oosten 2008). Poplar or willow crops not recognized as primary agricultural production, *i.e.* stands that exceed the 12-year rotation or stands that are not intensively-managed as a farm crop, can still qualify under the Managed Forestland class.

Another issue on the availability of land areas for biomass feedstocks is related to the assessment of the land suitability for afforestation of hybrid poplar trees for short rotation intensive cultures. A recent Canadian study (Joss *et al.* 2008) discussed the selection and magnitude of the environmental variables that can determine land suitability criteria, specifically for hybrid poplar.

Afforestation for production of biomass for biofuel – best practices

A number of published reports discuss the best practices for establishing plantations of fast growing trees for the purposes of biofuel production (UK DEFRA (Department for Environment, Food and Rural Affairs) 2004, van Oosten 2006, Isebrands 2007, Hansen *et al.* 1993, Hansen and Netzer 1985, White *et al.* 2010, Tubby and Armstrong 2002, Stanturf *et al.* 2001) , which include site preparation methods:

- deep ripping in conventional methods (ploughing and ripping have been suggested by most references as best practice in order to increase the fertility of the site for establishing of the new plantation trees),
- minimum tillage methods – disking, chisel plowing, sub-soiling, and mowing,
- on former forested areas – shearing, raking, piling, burning,
- use an orchard flail to reduce woody debris, followed by a rototiller, to further grind and incorporate debris into the soil,
- competition control/weed control (spray herbicide followed by shallow cultivation – the year prior to planting),
- weed competition must be controlled during the first growing season,
- tending (herbicides after the first growing season),
- fertilization (for example for Populus trichocarpa, at 4 years, to obtain 7-18 tons/ha/year, need 95-159 kg of N /ha/year),
- application of post-emergent herbicide such as glyphosate and Simazine followed by mechanical cultivation.

In terms of yield, crop density and crop cycle, the early yield projections did not consider potential negative impacts from diseases and insects (Dickmann 2006). Early trials with willows in Sweden and the US Midwest indicate that diseases, insects and abiotic events (such as frost in willows and wind damage in poplars) were the most important factors impacting yield. Any potential yield gains from intensive culture and improved genetic material will be undone by yield losses. Hybrid poplars were most affected by *Septoria musiva* stem cankers in eastern North America and willows by frost damage in the US Midwest trials (Hall *et al.* 1992) and Sweden. *Melampsora* leaf rust has been shown to affect poplar plantations in many regions of the world. General yield projections of short rotation woody crops have been lowered substantially from 20 – 34 ODT/ha/yr in 1991 (White *et al.* 1991) to 10 – 15 ODT/ha/yr (IEA Bioenergy Executive Committee 2002) in 2002-03 and 5 – 20 ODT/ha/yr in 2006, depending on material used, location and management intensity (Dickmann 2006). Hybrid poplar is reported in the literature to achieve harvest stages between 6 years in the US Pacific Northwest (DiPardo 2004) and 10-12 years in Canada (Hall 2002). Poplar growth and yield is discussed by Deckmyn *et al.* (2004) for short rotation coppice, Sartori *et al.* (2007) for ash application affects, Miller and Bender (2008) for poplar in the northeastern US, and Fang *et al.* (1999) for poplar hybrids with different density spacings. Hansen (1994) and Hansen and Netzer (1985) report on fertilizing poplar plantations. For Central European conditions: maximum MAI (mean annual increment) occurs at around 6-7 years for poplars, 10-12 years for aspen. Basic requirements include good water (minimum 350 mm rainfall during growing season) and nutrient supplies, deep soils and favourable climatic conditions (average air temperature at least 14 degrees Celsius between June and September). Weed control including herbicides is essential during establishing phase of poplar short rotation plantations (Kauter *et al.* 2003).

In terms of production systems, many alternative methods for the intensive production of hybrid poplar are possible. The major variables of alternative systems are spacing, rotation length, and cultural practices (including site preparation, weed control, irrigation, and fertilization). Spacing intervals have been proposed that range from 0.3 m by 0.3 m to 3.6 m by 3.6 m. Proposed rotations range from 4 to 15 years (Rose *et al.* 1981). Mechanized harvesting systems (similar to corn silage or sugarcane harvesters) can be used for short crop cycles in coppice systems. For hybrid poplar crops grown at longer crop cycles, in order to capitalize on improved average annual yields (mean annual increments), a more conventional system may be more appropriate (single- or multiple-stem tree harvesters). Lower planting densities (1000-2500 stems/ha) and longer rotations (8-12 years) promote greater diameter growth, and are preferable in cases where product flexibility is an objective, or where a high wood-bark ratio is important. Because improved genotypes usually are available at the termination of these longer rotations, replanting can be used rather than coppicing (Dickmann 2006).

Coppicing

Somerville *et al.* (2010) state that, in order to maximize the amount of woody biomass produced per hectare, the best practice appears to be to coppice harvesting, in which the plants are cut near the ground level after the end of the growing season every 3 to 5 years, depending on the species and the growing condition.

Earlier studies have suggested that the assumption that short-rotation coppice productivity increases from the first to second and subsequent rotations is not borne out in practice (Mitchell *et al.* 1999). The authors indicate that, in over half the experimental trials, the mean annual increment (MAI) in the second rotation was lower than the first. The conclusion that yields do not increase in the second rotation has been corroborated by other reports (Duffy and Beale 2005, Aylott *et al.* 2008).

Other studies have reported mixed results. Laureysens *et al.* (2005) looked at 17 clones hybrid poplar over two rotations; the best performers of the first rotation performed poorly in the second rotation, due to heavy rust infections; on the other hand, other clones showed low biomass production in the first rotation, but moderate to high biomass production in the second rotation. A later update on the same test site report mixed results of successive yields (Afas *et al.* 2008), indicating that, in the cases when biomass production decreased with each rotation, this may not be due to actual biomass accumulation in healthy trees (which would make sense to be increasing because they are taking advantage of a bigger and established root system), but instead to mortality or disease. In other words, the biomass/tree might be increasing with each rotation, but the biomass/hectare might be decreasing.

Fertilization

From the general best practices guides on short rotation plantations mentioned earlier, it seems that fertilization is expected to contribute to increased biomass yields. van Oosten

(2006) suggests 150-200 kg N per hectare, and 325-430 kg of urea fertilizer per hectare, at the start of the third (or fourth, fifth) growing season for the Canadian Prairie provinces, based on operational fertilizer experience. Isebrands (2007) state that best management practices for fertilization of poplars in Minnesota is annual applications of 56 kg/ha nitrogen (N) per acre. Stanturf *et al.*(2001) suggest typical fertilizer rates for Vancouver Island (non-irrigated) range between 25-200 kg/ha N at planting and canopy closure; for Pacific Northwest Eastside (irrigated) 60 kg/ha first year, increased by 30 every year to 150 by 4^{th} year. Bergusson (2010) suggests 67.25 kg/ha N in years 6, 8, and 10 of a 12-year single stem cycle, and 67.25 kg/ha N in years 6, 8, 10, 12, 14, 16, and 18 of an 18-year (3 rotations x 6 years each) coppice cycle.

However, besides the positive impact of fertilizers on biomass yield, using nitrogen-based fertilizers to grow crops destined for use as biofuels can incur large N_2O emissions (Revell et al. 2012)or even negate some of the benefits of displacing fossil fuels (Crutzen et al. 2007).

1.2.4 Land base impacts of biomass production

The land use and land-use change by forestry or agricultural activities, such as afforestation with short rotation tree species, leads to potentially substantial impacts on biodiversity and on soil quality. However, there is no widely accepted assessment method, thus far, for land use impacts (Canals *et al.* 2007). A recent IEA Bioenergy workshop (IEA Bioenergy Task 38 2009) suggests that calculation of the mitigation benefits of bioenergy must include emissions due to changes in soil organic carbon and biomass stocks resulting from direct land-use change. Much of the peer-reviewed literature on land-use change effects on GHGs for biofuel projects refers to the first-generation production of biofuels from mainly food crops (Fargione *et al.* 2008, Joslin and Schoenholtz 1997, Liska and Cassman 2008, Liska and Perrin 2009, Mathews and Tan 2009, Murray *et al.* 2003, Searchinger *et al.* 2008). The impacts of direct land-use change and the "carbon debt" have been discussed by Fargione *et al.* (2008), and the CO_2 released during the first 50 years after land conversion by Robertson *et al.* (2008). It is important to note that there are significant differences in the biofuels' costs and benefits reported in the literature, and the arguments that support one biofuel crop over another can easily change when one considers their full environmental effects (Scharlemann and Laurance 2008, Gutierrez and Ponti 2009).

1.2.5 Conversion technologies and biofuel products from wood feedstocks

Different technologies can be used to process woody biomass into fuel ethanol, such as biochemical conversion (BC), syngas-to-ethanol, thermochemical conversion (TC), consolidated bioprocessing (CBP), and gas turbine combined cycle (GTCC). Wood feedstocks can be used for the production of a variety of biofuels (Huber *et al.* 2006), and can also be combusted directly to provide electricity and process heat (Mann and Spath 2001, Demirbas 2003, Robinson *et al.* 2003). Wood can be converted to ethanol through enzymatic hydrolysis of the cellulosic fractions into sugars followed by fermentation of these sugars, with the lignin fractions being burned to provide heat and electricity (Hamelinck *et al.* 2005, Lynd *et al.* 2002, Lynd *et al.* 1991). The process is similar to the technology used in commercial biorefineries that convert corn to ethanol, however, the challenge with cellulosic ethanol is that hydrolysis of the cellulose is currently a difficult process that requires a pre-treatment to destabilize the lignin and hemicellulose, and make the cellulose accessible to cellulase enzymes.

Wood can also be gasified to produce hydrogen (Kumabe *et al.* 2007, Ptasinski *et al.* 2007), electricity, synthetic hydrocarbons such as gasoline and diesel through Fischer-Tropsch synthesis (Spath and Dayton 2003, Wang *et al.* 2005, Zwart and Boerrigter 2005), or other biofuels such as dimethyl ether (Semelsberger *et al.* 2006). Other valuable co-products may also be generated in the above-mentioned processes (Wyman 2003, Montgomery 2004, Ragauskas *et al.* 2006).

New technologies for producing biofuels from biomass are rapidly emerging, including the development of engineered yeast for increased ethanol yields (Alper *et al.* 2006), utilization of new microorganisms for ethanol production (Seo *et al.* 2004), pre-treatments for cellulosic digestion (Mosier *et al.* 2005), and fuel cells for converting sugars directly to electricity (Chaudhuri and Lovley 2003). Other bioenergy products from wood biomass include bio-oil, cellulignin, methanol, wood pellets, and biochar.

The production systems that process wood biomass into various energy products have been classified (IEA 2009) by their platforms, energy/products, feedstocks, and conversion processes, as shown in Figure 1.1 adapted from (IEA 2009):

- Main platforms – intermediates connecting different biorefinery systems and their processes – will include C5/C6 sugars, syngas, lignin, and pyrolitic liquid.
- Energy products: ethanol, electricity and heat, and synthetic biofuels.
- Two main feedstock groups: 'energy crops' from short rotation forestry, and 'biomass residues' from forestry, (bark, wood chips from forest residues, waste streams from biomass processing).
- Four main conversion processes: biochemical (fermentation); thermo-chemical (e.g., gasification, pyrolysis, combustion); chemical (e.g., acid hydrolysis, synthesis, esterification); and, mechanical processes (e.g., fractionation, pressing, size reduction).

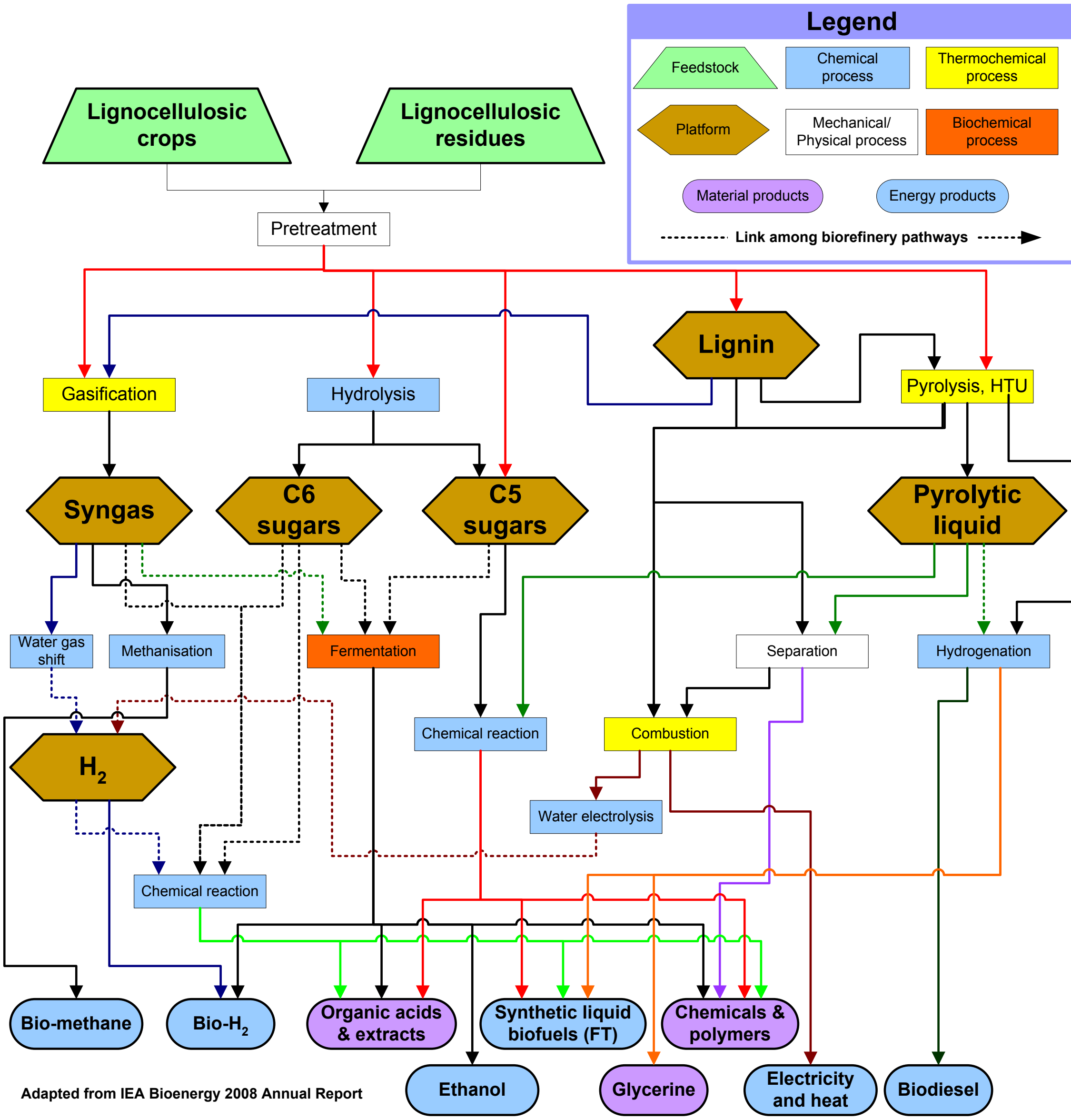

Figure 1.1. Overview of production platforms, energy/products, feedstocks, and conversion processes

Current commercial production of first-generation transportation fuels is transforming sugar and starch (*i.e.* from corn and sugar cane) in biofuel, using conventional technology. Even though they are regarded as a starting point in the development of sustained biofuel facilities, these first-generation biofuels such as corn ethanol have been reported to create carbon debts through land-use change (Fargione *et al.* 2008, Searchinger *et al.* 2008), to impact food supplies (Hill *et al.* 2006), to affect ecological damages caused by nitrogen and phosphorus fertilizers, pesticides, and erosion (Hill 2007, Robertson *et al.* 2008), and to have a high combined climate-change and health cost on the society (Hill *et al.* 2009).

Second-generation biorefineries are being developed on the basis of more sustainably-derived biomass feedstocks and cleaner thermochemical and biological conversion technologies, to efficiently produce a range of different energy carriers and marketable co-products: cellulosic ethanol, biohydrogen, biomethanol, DMF (2,5-Dimethylfuran), Bio-DME (dimethylether), Fischer-Tropsch diesel, biohydrogen diesel, mixed alcohols, and wood diesel. To avoid the criticism attributed to first-generation biorefineries, these new designs are aiming to reduce the impacts and maximize the benefits of social, economic, and environmental factors on a life cycle basis (IEA 2009).

The second-generation biofuels from lignocellulosic biomass (such as forestry and crop residues, corn stover, and switchgrass) are widely regarded as preferred feedstock for biofuel production because the vast abundance of biomass crops could support a larger biofuel industry than can be supported by food crops alone (Heiman and Solomon 2007). Cellulosic ethanol – if produced from low-input biomass grown on agriculturally marginal land or from waste biomass with minimal fertilizer, pesticide, and fossil energy inputs – has the potential to provide fuel supplies with greater environmental benefits than either petroleum or current food-based biofuels (Perlack *et al.* 2005, Hill *et al.* 2006, Hill 2007). Recent studies found higher environmental benefits (Hill *et al.* 2009) and lower combined climate-change and health costs (Hill *et al.* 2009) from cellulosic ethanol compared to corn ethanol.

However, current conversion processes for cellulosic biomass to biofuels are still under development, and large-scale harvesting, storage, and refinery systems are not yet cost-effective. There are no commercial-scale wood ethanol facilities operating anywhere in the world. Several companies operate pilot-scale facilities and are developing small commercial-scale biorefineries for wood chips, prairie grasses, and crop residues within a few years (Hahn-Hägerdal *et al.* 2006). Due to these current constraints, some authors predict that mature technology for large-scale deployment of cellulosic biofuels production will not be commercially available for at least a decade (Himmel *et al.* 2007).

Biochemical conversion is possibly the most mature process for the transformation of lignocellulosic materials into ethanol (Piccolo and Bezzo 2009). Many processes have been studied for biochemical conversion of lignocellulosic material to ethanol (enzymatic hydrolysis and fermentation). The dilute-acid pretreatment was referenced by many studies, with process variations: simultaneous saccharification and (co-)fermentation (Stone and Lynd 1995, Lynd et al. 1996, Wooley and Putsche 1996, So and Brown 1999, Wooley et al. 1999); consolidated bioprocessing (CBP) (Hamelinck *et al.* 2005); separate hydrolysis and fermentation (SHF) reported in Aden *et al.* (2002), process also used in Argonne's GREET model (Wang *et al.* 2007), in the GHGenius model ((S&T)2 Consultants Inc. 2003), and in Eggeman and Elander (2005). Huang *et al.* (2009) report techno-economic and engineering data for the conversion of hybrid poplar to ethanol.

1.2.5.1 Biofuel production technologies

Methodologies have been developed for calculating material and energy balances of conversion processes inside a biorefinery, using detailed equipment models to determine flow rates, composition and energy flow of all process streams. Examples are flowsheeting-type programs such as Aspen Plus (Spatari *et al.* 2010, Huang *et al.* 2009, Wu *et al.* 2006, Aden *et al.* 2002, Wooley *et al.* 1999, Wooley and Putsche 1996), HYSYS, and ChemCad.

As mentioned above, currently there are no commercial-scale wood ethanol facilities operating anywhere in the world, so evaluating the costs of production is still a theoretical exercise. Production of ethanol from cellulosic feedstocks is costly when compared to its production from starch based agricultural feedstocks (McAloon *et al.* 2000). Sassner *et al.* (2008) have assessed the cost effectiveness of three cellulosic feedstocks (salix, corn stover, and spruce) and they concluded that conversion technology used for ethanol production has more important implications for the cost-effectiveness of the conversion process than the type of feedstock used. Recently, Huang *et al.* (2009) found that for an ethanol mill based on simultaneous saccharification and co-fermentation technology, the ethanol production cost decreases with increasing plant sizes in the range of 1,000 dry Mg/day to 4,000 dry Mg/day. They also found that the cost of production of ethanol from hybrid popular increases if the plant size is more than 4,000 dry Mg/day as feedstock costs rise faster than non-feedstock costs. They estimated that the cost of ethanol production was not variable with the type of feedstock utilized *i.e.* corn stover, switch grass, hybrid popular, and aspen wood.

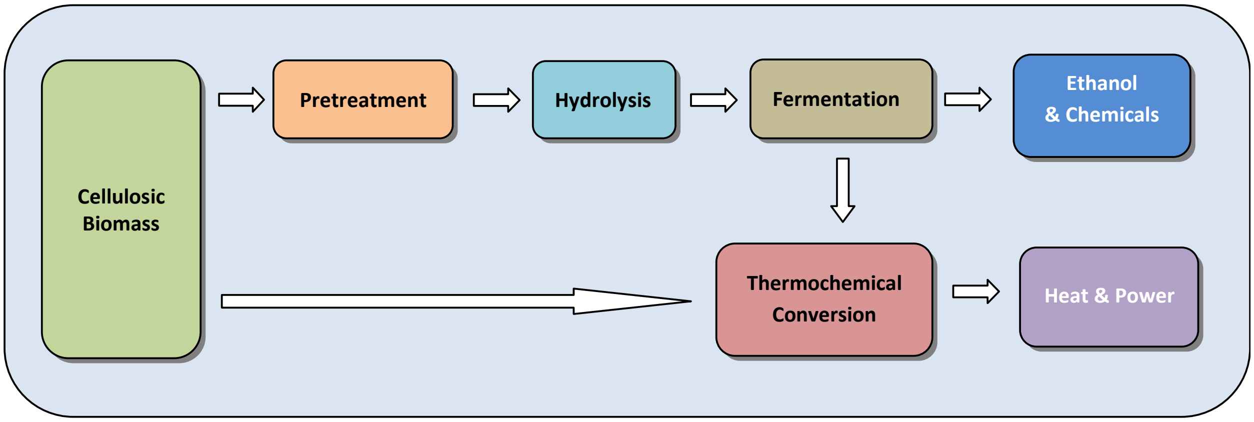

Figure 1.2. Schematic of the enzymatic hydrolysis conversion process

Published analyses have used hybrid poplar wood chips delivered at 50 wt% moisture to model forest resources (Spath *et al.* 2005). The design plant size reported by Aden *et al.* (2002) for a biochemical process was 2,000 dry tonne/day (2,205 dry ton/day). With an expected 8,406 operating hours per year (96% operating factor) the annual feedstock requirement was 700,000 dry tonne/yr (772,000 dry ton/yr).

Phillips (2007) modeled cellulosic ethanol production through gasification technology and catalytic conversion. The minimum selling price was found to be \$1.07/gal. Tembo *et al.* (2003) noted that the breakeven cost for the ethanol produced using thermochemical-fermentation technology will be about \$0.76/gal. Recently, Piccolo and Bezzo (2009) estimated that the cost of producing ethanol using gasification-fermentation based technology will be higher than that of enzymatic hydrolysis technology.

The Panel on Alternative Liquid Transportation Fuels with the National Research Council of the National Academies developed a model to simulate the capital and operating costs, and the carbon emissions of cellulosic ethanol plants using the SuperPro Designer chemical-process simulation software (National Research Council 2009). They considered poplar woodchips as their biomass feedstocks. The authors provide a sample detailed cost analysis for the "base-case" cellulosic ethanol production facility.

1.2.5.2 *Ethanol conversion efficiency*

The Panel on Alternative Liquid Transportation Fuels report (2009) assume 3 values for yield from poplar woodchips: low 67 gal/dry ton of biomass (BDT); medium 78 gal/BDT; high 87 gal/BDT. Huang *et al.* (2009) considered hybrid poplar among feedstocks with a conversion to ethanol efficiency of 88.2 gal/BDT. A frequently cited study (Wooley et al. 1999) used an ethanol conversion efficiency of 68 gal/ton, using yellow poplar as feedstock; it is noted that yellow poplar (*Liriodendron tulipifera*) is not a poplar but a relative of the magnolias.

1.2.6 Accounting methods for greenhouse gas emissions and carbon removals and sequestration

In forest ecosystems and tree plantations, carbon is stored in areas referred to as carbon pools or carbon stocks, which include above-ground biomass, below-ground biomass, minor vegetation, soil, and litter. Net changes in forest carbon stocks determine whether a forest ecosystem is a net source of atmospheric carbon or a net sink of atmospheric carbon. When a tree is harvested in traditional forestry operations, carbon is removed in the logs, but 40–60% of the tree biomass (branches, roots, leaves) remains in the forest where it decomposes slowly and gradually releases nutrients and CO_2. The harvested areas also regenerate so that over time a substantial new pool of carbon is created. In most Canadian forests, however, more carbon exists in soils and dead organic matter than in the living biomass (Greig and Bull 2009).

Carbon sequestration is measured in tonnes per hectare using carbon accounting models. Canada is developing a National Forest Carbon Monitoring, Accounting and Reporting System, which employs forest-inventory data, growth and yield information, and statistics on natural disturbances, management actions and land-use change to estimate forest carbon stocks, changes in carbon stocks, and emissions of non-CO_2 greenhouse gases. A key component of the system is the Carbon Budget Model of the Canadian Forest Sector, CBM-CFS (Apps *et al.* 1999, Kurz and Apps 2006).

The American US Forest Service carbon accounting model and measurement guidelines for the sequestration of forest carbon are discussed in (Smith *et al.* 2007, Pearson *et al.* 2007, Smith and Heath 2006). The carbon accounting rules and guidelines for the United States forest sector are presented in (Birdsey 2006). A discussion on measuring changes in ecosystem carbon in forestry-offset projects is given in (Hamburg 2000, LeBlanc 1999).

1.2.6.1 Accounting for CO_2 emissions from biomass

The timing of when CO_2 emissions occur compared to when CO_2 sequestration takes place has been reported as an important consideration in project carbon balance calculations (Rabl *et al.* 2007). However, many studies did not consider CO_2 in their analysis, treating bio-based CO_2 as being "carbon neutral", assuming that this balances with sequestrated carbon over the long term (Rafaschieri *et al.* 1999, Keoleian and Volk 2005, Matthews 2001, Fu *et al.* 2003, Heller *et al.* 2003, Heller *et al.* 2004, Forsberg 2000). These authors are assuming that CO_2 emissions need not be counted if emitted by biomass. One important issue that could be missed in such analyses is the benefit of adding carbon capture and sequestration (CCS) technology to the biofuel production facility, which would not be considered because that CO_2 is absent from the analysis.

To avoid such conclusions, Rabl *et al.* (2007) recommend that emission and removal of CO_2 be counted explicitly at each stage in the life cycle. In the example of a wood for bioelectricity, the sequestration of CO_2 should be counted explicitly for the biomass plantation, and the emission of CO_2 explicitly for the power plant. The net effect is, of course, zero or almost zero in this case: the biomass has been produced only to provide fuel for the power plant. If CCS technology would be considered at the power plant, such explicit accounting for carbon will automatically yield the appropriate results, whereas the above-mentioned carbon-neutral assumption would wrongly assume that removal and emission are balanced. Explicit accounting for CO_2 at each stage also allows the dynamic modeling of carbon emissions and removals. The time dimension is crucial for systems with a long delay between sequestration and emission of CO_2, such as the removal of biomass from long rotation forests stands, which will need decades to sequester back the carbon lost by harvesting. According to Rabl *et al.* (2007), it is not appropriate to neglect such delays, even if one does not use monetary valuation and discounting in quantifying the damage costs associated with climate change.

Materiality thresholds vs. de minimis emissions reporting

The literature concerned with carbon reporting and certification standards use the concepts of "materiality thresholds" and the "de minimis" emissions reporting. In general, accounting protocols and standards are meant for regulatory/compliance reporting purposes at the "company" level. This dissertation is analytical in nature, and the aim is to quantify GHG emissions along the value chain (from land-use change to final biofuel use), regardless to which "company" these activities belong. The rules for these reporting standards for regulatory/compliance purposes may not apply automatically to an analytical modeling approach.

The concept of materiality is drawn from financial reporting, where a material difference is sometimes taken to be an error or discrepancy of more than, say, 3% between reported and audited values. This potential discrepancy level is called the "materiality threshold", and it is about a potential error in reporting – but it is not directly related to the "de minimis" value for emissions, which is the permissible quantity of emissions that a company could omit from reporting in its inventory.

The GHG protocol of the World Resource Institute (2008) suggests that it is inappropriate to set a "one size fits all" materiality threshold. In order to utilize a materiality specification, the emissions from a particular source or activity would have to be quantified to ensure they were under the threshold. According to WRI, a *de minimis* threshold is not compatible with the completeness principle of their standard, and the materiality threshold should not be viewed as a *de minimis* for defining a complete inventory. They go further and recommend that companies need to make a good faith effort to provide a complete, accurate, and consistent accounting of their GHG emissions.

The Forest Carbon Standard Committee (2009) mention a total materiality threshold of 3% on the baseline inventory, which shall be met if the sum of all sources of errors does not exceed the threshold. This essentially means that each source of error needs to be evaluated, in order then to be able to evaluate the sum of all errors. The authors also discuss a *de minimis* emissions reporting value for each carbon pool as 5% of total increase in carbon stock. This provision could lead to unintended results when there are many carbon pools that decrease their carbon stock by about 5%, *i.e.* the sum of all these carbon decreases has the propensity to be a significant amount.

The CAR Forest Project Protocol Version 3.2 (2010) makes no mention of *de minimis* values; materiality thresholds are mentioned only in the context of reporting errors, as above. The EPA Climate Leaders / National Greenhouse Gas Inventory Protocol (US EPA 2008) corroborate the recommendations of the WRI GHG Protocol that materiality threshold should not be viewed as a *de minimis* for defining a complete inventory.

1.2.6.2 Amount of existing carbon stocks that are present on land that will be converted to fast growing tree plantations

Two recent reports by Tyner *et al.* (2010) and Searchinger *et al.* (2008) use the same numbers for carbon stocks in Canada. For carbon in vegetation: 160 tonnes C/ha for temperate evergreen forest; 135 tonnes C/ha for temperate deciduous forest; 7 tonnes C/ha for temperate grassland. For carbon in soil: 134 tonnes C/ha for both temperate evergreen forest and temperate deciduous forest; 189 tonnes C/ha for temperate grassland.

The IPCC Guidelines for National GHG Inventories (2006) use a range of 20-130 tonnes C/ha for soil C stocks under native vegetation, depending on the type of soil, for "cold temperate, dry" and "cold temperate, moist" (chapter 2, table 2.3, page 2.31); IPCC seems to also apply these numbers for cropland. (Note: the IPCC numbers for soil C are for 0-30 cm depth). However, the study that is referenced by IPCC, Jobbagi and Jackson (2000), report (table 3, page 430; for 0-100 cm depth) 112 tonnes C/ha in soil organic carbon for croplands; 174 tonnes C/ha for temperate deciduous forest; 145 tonnes C/ha for temperate evergreen forest; and, 117 tonnes C/ha for temperate grassland.

Two BC-specific reports mention carbon stocks data for forestlands. Fredeen *et al.* (2005) report for 2nd growth in BC Interior forestlands: 35-41 tonnes C/ha in vegetation; 78-83 tonnes C/ha (0-47 cm) and 106-112 tonnes C/ha (0-106 cm) in soil; 27-29 tonnes C/ha in forest floor; 9-13 tonnes C/ha in woody debris. Shaw *et al.* (2005) report on C stocks in the Montane Cordillera of BC (numbers adapted from Fig. 11; results for 2nd growth): 43-226 tonnes C/ha in soil, and 42-74 tonnes C/ha in tree biomass.

1.2.6.3 *Potential carbon losses in soil and vegetation from initial direct land-use-change*

Gershenson *et al.* (2009) report findings from a comprehensive review on soil carbon changes following land-use change and establishment of tree plantations or afforestation projects. The authors corroborate findings of other published research that high disturbance site preparation activities, such as plowing, deep ripping, etc. will have significant negative effects on soil carbon, with potential losses as high as 30%. According to their review, the soil carbon lost during harvest activities is recovered in some systems within 50 years, but the interval is longer for more northern, less productive systems, and can be more than 100 years in some cases; this effect is dependent on soil type. Since initial losses from harvest activities can be as high as 20% of ecosystem carbon, the authors suggest that an inter-harvest period of adequate length is critical for ensuring that such losses are replenished (unlike short rotation plantations, which have short intervals between successive harvests, *i.e.* a few years). The authors suggest that one of the critical variables in the effects of harvests on soil carbon is time; soil carbon stocks generally recover with time after harvests, although the recovery time is greatly dependent on subsequent forest productivity. Gershenson *et al.* mention the recent recognition of soil carbon importance within the Clean Development Mechanism of the Kyoto Protocol (CDM) Afforestation/Reforestation guidelines; these guidelines specify that, in order to ensure soil carbon stability, physical disturbance should not exceed 10% of the project area, woody debris from harvesting should be left on-site, and removal of existing vegetation as part of site preparation shall not constitute more than 10% of the project area, with some caveats for traditional management. Gershenson *et al.* critiqued the meta analysis of 73 studies by Johnson and Curtis (2001) on effects of different harvesting techniques on soil carbon, pointing out that the meta analysis was based on averaging the effects of reported studies and is not very useful for policy recommendations, as it includes widely different studies with different harvesting techniques and in different biomes. Gershenson *et al.* continue to suggest that the overall results of the majority of studies in the meta analysis showed that, after harvest, ecosystems can experience anywhere between 30% soil carbon loss and a 60% soil carbon gain, which roughly translates to a 15-20% ecosystem carbon loss to a 30-40% ecosystem carbon gain; the overall weight of evidence from the Johnson and Curtis (2001) review points to the importance of retaining residues on site; however, there are some studies that disagree with this conclusion. Gershenson *et al.* contend that multiple studies suggest that rotation length, rather than harvest intensity, is the major factor driving the effects of harvesting practices on soil carbon, citing Seely *et al.* (2002) whose conclusions concur with the above, that in all cases longer rotation lengths had a positive effect on overall soil carbon, and that soil carbon accumulation was most likely expected in rotations over 50 years for aspen and pine, and over 100 years for spruce, a species with much slower stand development. Seely *et al.* also found that intervals between rotations shorter than 50 years resulted in 10%-20% losses in soil carbon regardless of tree type. Gershenson *et al.* further reference Nave *et al.* (2010) who found an effect of rotation length, but also found that soil type significantly affects the magnitude of this effect, with the average recovery time (return of soil carbon to pre-harvest levels) in some soils approaching 80 years, while data from the other soil orders is inconclusive due to lack of long-term studies. The authors conclude that available evidence suggests that rotation intervals that are less than 50 years may not be sufficient to replace soil carbon lost during prior harvests (which can be as much as 60% soil carbon, 30% ecosystem carbon for hardwood forests growing on Alfisols-type soils).

Other studies make simplistic assumptions about soil carbon dynamics in biofuel plantations, possibly recognizing that accurate projections of changes in soil organic matter are limited by the fact that our current understanding of soil organic matter dynamics is incomplete, and the exact influence of any given factor on soil organic carbon dynamics is poorly understood. For example, van Kooten *et al.* (1999, 2000) assume a linear constant yearly accumulation of C in soil of 0.96 tonnes C/ha/year (up to a maximum of 48 tonnes/ha attained after 50 years) for afforestation with successive 15-year rotations of hybrid poplar. Their assumption was based on a report by Guy and Benowicz (1998), who actually assumed a 55 year period needed to regain carbon stores and reach equilibrium after planting, citing in turn a study by Birdsey (1992). However, the assumption that once an agricultural land is converted to a forest it takes 55 years for the soil C stocks to grow and reach equilibrium, is problematic when applied to short rotation plantations, because 1) the afforested area is highly unlikely to be left undisturbed for 55 years, and 2) Birdsey seems to make no such claims about dynamics of soil carbon accumulation. It is interesting to note that Guy and Benowicz caution that their projections are very tentative, and acknowledge that a small initial decrease in carbon content may be observed after planting due to the soil disturbance.

Birdsey (1992) discussed carbon storage in forest soils in the US, however, the study only described the amount of soil carbon in forests at some point in time; it did not study the soil carbon rate of accumulation over time. Birdsey's analysis also did not refer to afforestation of agricultural lands, and did not consider the forests as being "mature". Moreover, Birdsey does not mention any "equilibrium" level for soil C.

McKenney *et al.* (2004) use the same algorithm as van Kooten *et al.* (1999) to represent carbon pools, as well as using the same "linear accumulation" of carbon in soil at the rate of 0.96 tonnes C/ha/year for 50 years. McKenney *et al.* (2006) did not consider any post-clear-cut fluctuations of soil carbon content, although they acknowledge that small decreases in carbon content in soils may be observed after harvesting, citing Hansen (1993).

Hansen *et al.* (1993) report that poplar plantations between 4-6 years old may actually lose soil carbon, and establishing and tending plantations often results in early soil carbon loss. They suggest that soil carbon loss under trees occurred most frequently from the surface 30 cm early in the plantation history.

Samson *et al.* (1999) studied the rate of soil organic carbon change in short-rotation plantations compared with adjacent agricultural crops, and reported that "the longer the rotation length, the greater the soil carbon storage potential. Very short rotations (less than 10 years) may therefore not lead to soil carbon sequestration. In fact, carbon is most probably lost during the establishment phase of the plantations".

Grigal and Berguson (1998) concluded that soil carbon was most probably lost during the establishment phase of the plantations causing sharp decreases in soil carbon, agreeing with Hansen (1993) that soil C is likely to be lost during the initial years of plantation establishment.

IPCC Guidelines for National GHG Inventories (2006) state that conversions on mineral soils generally either maintain similar levels of C storage or create conditions that increase soil C stocks, particularly if the land was previously managed for annual crop production.

However, under certain circumstances, grassland conversion to forest land has been shown to cause C losses in mineral soils for several decades following conversion (Paul *et al.* 2002). The CAR Forest Project Protocol Version 3.2 (2010) states that site preparation activities that involve deep ripping, furrowing, or plowing where soil disturbance exceeds 25 percent of the project area have been identified as having the potential to result in significant changes in soil carbon, and therefore are deemed relevant carbon pools that need to be monitored and reported on.

Guo and Gifford (2002) report in their meta-analysis that soil C of younger broadleaf plantation forests decreased greatly: -25% for plantations < 20 years, -22% for 20-40 years, while soil C stocks somewhat recovered in plantations over 40 years old (+2%). Land-use change from pasture to broadleaf plantation somewhat decreased the soil C stocks (-3%). The authors report that conversion of native forest and pasture to plantation had little effect on soil C stocks in the lower rainfall (<1200 mm) areas, but significantly reduced soil C stocks in higher rainfall areas, especially in the areas with precipitation > 1500 mm (-23% reduction in soil C stocks).

Berthrong *et al.* (2009) report that soil carbon decreased 6.9% when trees were grown on grasslands, pasture, or shrublands. Paul *et al.* (2003) found that soil carbon is typically lost in the first decade after forest establishment, but most forests eventually recover most of the lost soil carbon after 30 years. Tyner *et al.* (2010) and Searchinger *et al.* (2008) assumed 25% loss of carbon in soil organic carbon in the top meter, and 100% loss of carbon in vegetation, resulting from direct land use conversion.

Delucchi (2011) suggests that changes in land use affect the oxidation and formation of carbon in plants and soils, and the start of cultivation of a biofuel crop creates three streams of carbon emission or sequestration: (1) the combustion or decay of the original, native ecosystem plant biomass, (2) a change in the carbon content of the soil, and (3) the growth/harvest cycles of the biofuel crop.

Two recent reports on soil carbon dynamics looked specifically at hybrid poplar (*P. Euramericana*) afforestation of marginal or former agricultural land, monitored up to 20 and 15 years of age respectively (Mao *et al.* 2010, Zhang *et al.* 2010). In both studies, organic C stocks in the soil have decreased in the first 10 years (8, respectively) following afforestation. Mao *et al.* state that forestry operations during pruning generally lead to soil compaction (pruning was done at 5 years of age), and that the initial decline in soil C stock was due to the relatively little organic C input through plant residues, the continued decomposition of residues of the preceding agricultural phase, and accelerated mineralization of organic matter induced by forest operation during site preparation and pruning.

In conclusion, the surveyed literature increasingly suggests that afforestation for biofuel projects could result in potentially significant biogenic carbon impacts, due to the large proportion of organic matter stocks from the total carbon stored in the land base of the project. However, one question that remains unanswered is what impact the biogenic carbon dynamics may have on not only the carbon balance of a biofuel project, but on the total net GHG balance of the project.

1.2.6.4 Carbon losses in soil at harvest times from harvest-regeneration operations

Nave *et al.* (2010) report 9% losses in mineral soil C following harvest (page 861), and 20% losses in surface mineral soil C when tillage is used following harvest (page 863). They also mention 36% losses in forest floor C at harvest-regeneration time (Fig. 2 and page 860). Gershenson *et al.* (2009) suggest that one of the results of harvest operations is mechanical disturbance to the forest floor, which acts on soil carbon in various ways, from 24% (Huntington and Ryan 1990) to as much as 92% (Martin 1988) of the forest floor disturbed as a result of harvest operations. Citing Schmidt *et al.* (1996), the authors suggest that some of this disturbance is an intentional part of site preparation, such as disking or plowing, and results in significant losses: over 20% of soil carbon, and 10-15% of total ecosystem carbon. Gershenson *et al.* further state that studies exist that examine the direct effect of plowing on soil carbon in forests, but data from agricultural systems show that plowing has an immediate negative effect on soil carbon, with losses from only a few years amounting to 30% of the total carbon pool, and restoration studies have shown that, without intensive management, this carbon is difficult to get back, citing Ammann *et al.* (2007). The authors suggest that some of this disturbance is incidental to harvest operations, and is a result of the use of heavy machinery, which disturbs the forest floor and often results in mixing the top soil horizons. Such high levels of disturbance break up the forest floor carbon layer and incorporate it into mineral soil, as well as expose mineral soil to the atmosphere, causing carbon to be oxidized and emitted as CO_2. Gershenson *et al.* add that hydrologic processes can cause carbon to be lost through erosion, and physically re-arrange the forest floor and mineral soil, increasing the difficulty of tracking carbon losses, citing Black *et al.* (1995).

1.2.6.5 Biofuel sustainability standards

There are currently no industry-wide biofuel "sustainability" standards. Various standards for sustainable production that account for land use and other environmental impacts, social impacts, and effects on GHG emissions are being developed. For example, the Forest Carbon Standards Committee, a Canadian-American initiative – organized by American Forest and Paper Association, Forest Products Association of Canada, Society of American Foresters, and Canadian Institute of Forestry – aims to develop a bi-national standard to measure carbon from forestry activities that is environmentally sound, scientifically based, and economically feasible (Forest Carbon Standards Committee 2009). This standard development process is accredited by the American National Standards Institute (ANSI) and the Standards Council of Canada (SCC).

Accounting for carbon is becoming a major undertaking for carbon emissions reporting by businesses in the supply chain (PricewaterhouseCoopers 2009), some being concerned only with their "carbon footprint" (Wiedmann and Minx 2007). There is much discussion on the emerging bioenergy markets (Verdonk *et al.* 2007), and the regulatory framework for a sustainable bioenergy policy and the setting of sustainability standards (WBGU 2008).

Another initiative, the Roundtable on Sustainable Biofuels at the École Polytechnique Fédérale de Lausanne, is also leading an international multi-stakeholder effort in the development of bioenergy standards (Roundtable on Sustainable Biofuels 2010). Many other authors and organizations discuss the emerging sustainable standards for bioenergy (Fritsche *et al.* 2006, Roundtable on Sustainable Biofuels 2010, Ingerson and Loya 2008, Olander 2008, Sampson *et al.* 2007, Fritsche *et al.* 2006, van Dam *et al.* 2008, Lewandowski and Faaij 2006, Dehue *et al.* 2007), and the current literature reflects the on-going debate on carbon accounting related issues (Birdsey 2006, Cathcart and Delaney 2006, Greig and Bull 2009, Hamburg 2000, Kim *et al.* 2008, Kurz and Apps 2006, LeBlanc

1999, Matthews *et al.* 2008, McKenney *et al.* 2004, Miner 2006, Pearson *et al.* 2007, Rabl *et al.* 2007, Smith *et al.* 2007, von Blottnitz and Curran 2007, Wise *et al.* 2009, Broekoff and Zyla 2008, Ingerson and Loya 2008, Ruddell *et al.* 2007, Werner and Nebel 2007).

1.3 Problem definition

Projects that aim to establish fast growing tree plantations and substitute gasoline with ethanol from the resulting wood biomass have the potential to reduce atmospheric carbon dioxide and other greenhouse gas stocks, and to increase terrestrial carbon stocks – compared with business as usual (BAU), *i.e.* what would have happened in the absence of the biofuel project – through one or more of the following situations:

1) absorbing more carbon dioxide from the atmosphere through biomass growth than BAU;
2) increasing terrestrial carbon stocks through sequestration of carbon in soil organic matter during the project and in live biomass that is not harvested by the end of the project, more than BAU;
3) emitting less carbon dioxide than BAU through natural decomposition of soil organic matter; and/or,
4) producing a lower net greenhouse gas emissions balance than that of the displaced fossil fuel production system (*i.e.* less than BAU).

However, as discussed in the previous section, recent studies indicate that:

- terrestrial carbon stocks may not necessarily increase as a result of establishment of fast growing tree plantations (for example when high initial carbon stocks are reduced through direct land-use change, but also when the frequent harvest/planting operations hinder the accumulation of carbon in soil),
- emissions through natural decomposition of soil organic matter can be significant and may in fact increase as a result of the biofuel project (when the initial organic mater stocks are large and the mechanized activities disturb the soil and release carbon),
- net carbon and greenhouse gas balance dynamics do change with time, throughout the project time horizon.

Current methodologies for evaluating the viability of biofuel projects to decrease atmospheric carbon dioxide and other greenhouse gases do not consider specifically the impact of all these issues (e.g., the impact of biogenic carbon dynamics and emissions from decomposition of organic matter to the net balance of project greenhouse gas emissions) in a time-dependent manner and from a life cycle and project-level perspective.

This dissertation aims to address the knowledge gap of quantifying the impact of changes in organic matter stocks of afforestation-for-biofuel production systems – encompassing the full cycle from land-use change and biomass production to the final use of the biofuel product – in terms of the potential of these projects to reduce atmospheric carbon dioxide and other greenhouse gas stocks and increase terrestrial carbon stocks, compared with the displaced fossil fuel system.

1.4 Research questions and objectives

Following from the problem statement presented above, the key research question of this dissertation is:

Under what conditions can a biomass-to-biofuel production system reduce the atmospheric carbon dioxide and other greenhouse gas stocks and increase the terrestrial carbon stocks, relative to a baseline determined by the displaced fossil fuel production system, when considering the potential impact of initial carbon and organic matter stocks, and of emissions from dead organic matter decay, in a time-dependent framework and from a life cycle and project-level perspective?

To enable answering the research question above, the research objectives of this dissertation are as follows:

1. Develop a biomass production planning model for projects using fast growing tree plantations in such a way as to allow the subsequent quantification and monitoring of organic matter stocks dynamics and greenhouse gas fluxes, in a time-dependent manner and from a life cycle perspective. This model will also determine the plantation area needed and the ethanol production cost (addressed in CHAPTER 2).

2. Develop a biogenic carbon accounting model, using as input the planning information from the above-mentioned biomass production model, which is able to quantify and monitor throughout the project planning horizon: (a) the removal of carbon dioxide from the atmosphere by growing biomass, (b) the initial carbon stocks after land-use change, (c) the transfer of carbon between live and dead organic matter pools, both above- and below-ground, (d) the removal of carbon from site in harvested biomass, and (e) the release of carbon to the atmosphere through natural processes of organic matter decomposition (addressed in CHAPTER 3, building on the model developed in CHAPTER 2).
3. Develop a model for calculating the net greenhouse gas balance of biofuel projects on a life cycle basis (from the initial land-use change, establishment of plantations and construction of biorefinery, through the conversion of biomass into ethanol and the final use in internal combustion engines) throughout the project time horizon, including biogenic carbon as well as all other greenhouse gas emissions generated by the project, and any additional credits for substituting fossil fuels (addressed in CHAPTER 4, incorporating the models developed in CHAPTER 2 and CHAPTER 3).
4. Using these models, identify the conditions under which the biofuel production system can be a viable climate mitigation strategy, *i.e.* when it increases the terrestrial carbon stocks and reduces the greenhouse gas emissions relative to a baseline determined by the comparable fossil fuel production system displaced. This objective also includes identifying the conditions when the biofuel project is not a viable strategy (addressed in CHAPTER 4).
5. Determine the impact and relative importance of input variables (for the model developed in CHAPTER 4) on the biofuel project net greenhouse gas balance for several project time horizons, as well as on plantation area and on ethanol production cost (addressed in CHAPTER 5).

1.5 Structure of dissertation

This document follows the structure of The University of British Columbia's (2013) guidelines for structures of theses and dissertations. The following sections will form the main body of the dissertation:

- CHAPTER 2: Modeling biomass-to-biofuel projects from short rotation tree plantations with the Carbon-aware Biomass Production Optimization System (C-BOS);
- CHAPTER 3: Modeling the life-cycle biogenic carbon balance of afforestation-to-biofuel projects with the Biogenic Carbon Dynamics model (Bio-CarbD);
- CHAPTER 4: C3BO: a method for assessing the greenhouse gas and carbon balance of biofuel projects that displace gasoline with wood ethanol from fast growing tree plantations;
- CHAPTER 5: Wood-to-ethanol bioenergy from fast growing tree plantations: a sensitivity analysis of project net greenhouse gas balance, ethanol production costs, and plantation area.

The conclusion of the dissertation (CHAPTER 6) will present an overall analysis of the research chapters findings, discuss the outcomes of the dissertation in light of current research in the field, and suggest potential further research directions based on the dissertation work.

CHAPTER 2 Modeling biomass-to-biofuel projects from short rotation tree plantations with the Carbon-aware Biomass Production Optimization System (C-BOS)

2.1 Synopsis

This chapter describes the development and implementation of the Carbon-aware Biomass production Optimization System (C-BOS), a biomass production planning model for analysis of biomass-to-biofuel systems. This model is designed for subsequent integration with a carbon accounting model and a life-cycle biomass-to-biofuel GHG balance model, as decision-support tools for assessing the viability of displacing gasoline with ethanol from wood as a climate mitigation strategy. C-BOS models the biomass-to-biofuel production system such that all the carbon-related impacts (*i.e.* sequestration and emissions) on biomass stocks and on the landscape can be subsequently quantified over a set time horizon. This is accomplished through maintaining land parcels (harvest decision units) intact over time, and monitoring all necessary live biomass pools (*i.e.* stem, bark, branches, foliage, coarse roots, fine roots) on each land parcel. The utility of this model is shown on a test case for short rotation poplar plantations in the Pacific Northwest, by considering potential future gains in biomass growth yield and conversion to ethanol efficiency, biomass production options (treatments), transportation distances, and land productivity types.

2.2 Introduction

As mentioned previously, developing sustainable alternatives to fossil-derived transportation fuels is an important component of a climate change mitigation strategy. Substituting fossil transportation fuels with renewable biofuels, such as ethanol produced via biochemical conversion from wood biomass grown in short rotation poplar plantations, is viewed as a potentially viable alternative (National Research Council 2009).

A typical fuel-switch project that aims to substitute a defined quantity of gasoline with an equivalent (either volumetrically or energetically) amount of ethanol over a specific time horizon, would be comprised of:

a. the establishment of fast growing tree plantations that would produce the necessary wood biomass,

b. the construction and operation of a biorefinery specially designed and constructed for this purpose, which would consume the wood biomass originating from tree plantations,

c. the conception and operation of all the necessary production activities of the biomass-to-biofuel supply chain, including land-use change, transportation of biomass to biorefinery, and transportation of ethanol to fuel blending stations.

From the biofuel perspective, a key question that arises is: under what conditions can a biomass-to-biofuel production system be a viable climate mitigation strategy? More specifically, are there scenarios that increase the stored carbon stocks and/or reduce the greenhouse gas emissions relative to a baseline determined by the displaced fossil fuel production system? Three fundamental modeling elements are necessary to answer this question:

i. modeling appropriately the activities of the short rotation biomass production system over a time horizon of many decades into the future, in such a way that all the carbon-related impacts (*i.e.* sequestration and emissions) on biomass stocks and on the landscape can be subsequently quantified over time,

ii. modeling the dynamics of all the biogenic carbon stocks and fluxes within the biofuel production project, including carbon removals from the atmosphere through natural growth processes of above- and below-ground biomass, the carbon transfers between live biomass pools and dead organic matter (DOM) pools, and the carbon emissions from biodegradation of organic material through both DOM decay and biomass to biofuel conversion processes,

iii. modeling the life-cycle carbon balance of the biofuel production system (including impacts from land-use change, emissions from using fossil fuels for all biomass- and biofuel-production processes) and comparing it with that of the displaced gasoline production system. While element (iii) would be able to eventually answer the mitigation viability question, it is dependent on the suitable monitoring of biomass and carbon pools by element (ii), which in turn is dependent on an appropriate model being developed for biomass production by element (i).

This chapter is concerned only with the modeling framework for element (i): modeling the short rotation biomass production system with a tactical forest planning model, in such a way that all the carbon-related impacts (*i.e.* sequestration and emissions) on biomass stocks and on the landscape can be subsequently quantified over time. The model for element (ii) is described in CHAPTER 3, and element (iii) in CHAPTER 4.

A typical approach to tactical forest planning is to employ mathematical models that are able to predict the medium-term outlook of different forest management inputs. One type of such modeling frameworks is mathematical programming in general, and linear programming in particular. The linear programming planning models explicitly investigate the choice of forest growth and management (planting and harvest scheduling) strategies across the landscape, while ensuring the sustainability of forest growth (for biomass quantity objectives), and the viability of the utility to the forest owner (for net revenue financial objectives) (Gunn 2007).

Two of the most well-known modeling approaches to forest growth and management are the Model I and Model II of Johnson and Scheurman (1977); see also Davis *et al.* (2001). The two algorithms represent techniques for prescribing optimal forest harvest activities and investment under different objectives, in a multi-period linear programming structure. This makes both Model I and Model II suitable for consideration as a modeling structure for the planting and harvest scheduling activities of the biofuel production system considered in this chapter. Gunn (2007) offers a succinct discussion of the two modeling approaches. The Model I formulation accommodates the need for spatial representation, as harvest areas (individual land parcels) are kept intact over time. In Model II, all stands of the same age class are aggregated, so individual land parcels are not kept intact through time. In practice the arcs of Model II are usually only a small fraction of all possible paths. Thus, Model II appears to be a more efficient modeling framework. However, because Model II merges land parcels at harvest, validly representing growth requires a separate network for every different site capability and land type (Davis *et al.* 2001). To account for different management regions and biophysical zones, and to keep track of the spatial land parcels through time (for carbon stocks accounting, for example), separate Model II networks are required for each unique combination of attributes. This can result in very large LP models with substantial network constraints. Such models can be relatively difficult to solve because of the degeneracy caused by the network constraints.

In Canada, REMSOFT developed the WoodstockTM/StanleyTM based on a hierarchical approach described by Jamnick and Walters (1993), based on a Model II timber supply model structure. Woodstock includes a module that can interact with the Canadian Carbon Budget Model CBM-CFS3 (Kurz *et al.* 2009) in order to include the dynamics of live biomass carbon pools. Woodstock first solves the harvest scheduling problem independently of the carbon pools, and the resulting merchantable volume yield data are *a posteriori* input into the CBM-CFS3, which then generates carbon yield tables. An important issue is that in Model II the forest age classes are collapsed after harvest, and individual land parcels are not being kept intact over time. When land parcels are being grouped together based on their harvest ages (and implicitly on the merchantable stem carbon pool), all other carbon pools are also grouped together. However, on non-homogenous landscapes (*i.e.* with different soil and DOM carbon stocks) it is unlikely that these carbon pools would have the same amounts of stored carbon across the land parcels being grouped, and therefore this information is "lost" for future iterations. This "loss of history" for the carbon pools stocks and flows makes it impracticable to accurately track carbon stocks and dynamics through time.

Neilson *et al.* (2006) investigated the modeling of carbon sequestration with CO2Fix (Masera *et al.* 2003, Schelhaas *et al.* 2004) and Woodstock. They used CO2Fix to simulate the carbon pools yields, then combined the simulated pools into one C yield, which was then included in the Woodstock formulation to generate a forest harvest plan. Neilson *et al.* recognized the limitations of this approach (*i.e.* using a Model Type II): the application of C yield curves in Woodstock is unlikely to fully conserve C (Neilson *et al.* 2006).

Hennigar *et al.* (2008) used stand projections of merchantable volumes of timber obtained from WoodStock as input into CBM-CFS3 to generate stand-level carbon yields which they grouped in only four carbon pools; however, the slow to very slow decaying DOM pools were not considered in their model simulations. Neilson *et al.* (2008) also used WoodStock and CBM-CFS3 to investigate carbon storage in forest and wood products. Similar to the approaches of Neilson *et al.* (2006, 2007) and Hennigar *et al.* (2008), the dynamics of biomass and DOM carbon stocks are obtained *a priori* from CBM-CFS3, and then the C yields of merchantable stem biomass (considered as the equivalent of timber yields) are used as inputs in WoodStock.

From a carbon monitoring/accounting perspective by integrating the CBM-CFS3 with a Type II timber supply model, there are three potential issues with this approach. Firstly, while the merchantable stemwood pool has slow turnover rates (0.45–0.67 %C/yr), the branches pool have faster turnover rates of (3-4 %C/yr), and the foliage even faster (95 %C/yr) in the CBM-CFS3 (Kurz *et al.* 2009); it is not clear what turnover rate was used for the combined live biomass pool. CBM-CFS3 has 5 live biomass pools and 9 DOM pools, which have different turnover and decay rates. Secondly, the DOM pools that are receiving this turnover are different for each live biomass pool; it is not evident what is the DOM pool that receives the turnover from the combined live biomass pool. Thirdly, the base decay rates of the DOM pools are different (1.87%/yr for snag stems, 7.18%/yr for snag branches, 14.35%/yr for AG fast, and 35.5%/yr for AG very fast).

It is reasonable to expect then that the grouping of carbon pools that have different turnover, litterfall transfer, and decay rates – an artefact of using the Model Type II formulation – may result in incorrect representations of C stocks dynamics over time, especially when these pools contain a large amount of carbon stock. To address the limitations of integrating carbon accounting with a Model II formulation, this chapter proposes a biomass production planning model with a Model I structure instead. We describe the development and implementation of the Carbon-aware Biomass Production Optimization System (C-BOS), a tactical production planning model that determines the optimal biomass production strategy, and that is capable of representing the land parcels and biomass/organic matter pools in such a way that they can be subsequently quantified for a life cycle carbon balance analysis. Specifically, the methodology proposed in this study quantifies the biomass pools and the respective biogenic carbon pools on land parcels that are kept contiguous through time. This permits the model to monitor not only the harvested stem biomass, but all the other live biomass pools (bark, branch, foliage, coarse roots, fine roots). This also allows for transportation distances (and associated costs, diesel consumption, etc.) to be linked to specific land units, which can be of benefit to subsequent analysis of the GHG impact of emissions from harvested biomass transportation.

As a practical implementation of the proposed model, we developed test case scenarios for a biomass-to-biofuel production system with fast growing poplar trees (rotation ages of fifteen years or less) in the Pacific Northwest. It is assumed that the land areas chosen for biomass production had a different use before the biofuel project (e.g., unused or unmanaged marginal land), and that biomass is produced with the only goal to convert it into liquid biofuels in one biorefinery facility (*i.e.* biomass is not to be used/sold outside of the biofuel project). The biofuel project consists of: 1) the construction of a biorefinery that would produce an annual amount of biofuels from wood biomass feedstock; 2) the conversion of land areas to plantations of fast growing trees, sufficient to supply the necessary annual wood biomass in the quantity demanded by the biorefinery; 3) the production activities of wood biomass through planting and harvesting cycles; 4) the transportation of biomass to the biorefinery; 5) the conversion of biomass into biofuels at the biorefinery; and, 6) the transport of biofuels to fuel-blending stations. The analysis of the proposed biomass-to-biofuel production system is done at the project-level, meaning that all the land use, production activities and their costs are accounted for their contribution to the biofuel project.

The remainder of the chapter is organized as follows. Section 2.3 describes the methodology. Section 2.3.2 presents in detail the test cases analyzed and the modeling assumptions. The results are shown and discussed in Section 0 along with a consideration of the model limitations. Concluding remarks are presented in Section 2.5.

2.3 Methodology

The carbon-aware biomass production optimization planning system (C-BOS) proposed in this chapter is based on a modified version of a classical harvest scheduling optimization model (Type I) using a linear programming formulation (Johnson and Scheurman 1977). C-BOS simulates harvest-regeneration activities on spatially-located land management units, considering multiple possible soil capability classes, tree species and associated growth and yield curves and rotation ages, biomass production activities, and transportation distances to biorefinery. The objective function of the linear programming model is to minimize project costs, while producing a specified annual volume of biofuel from biomass.

The model is described mathematically below. The equations are explained in the section following the formulation.

2.3.1 Formulation of the model

To describe the model a rather large number of variables (*i.e.* several thousand, depending on the number of land units, treatments, and project horizon) was used, according to a uniform notation scheme, as follows:

$E_{i_1 \ldots i_n}^{(a)}$

where,

E is the entity. ***LU*** :: land unit; ***P*** :: land unit parcel; ***T*** :: parcel treatment

$\boldsymbol{i_1} \ldots \boldsymbol{i_n}$ is a generic index, and

a is an attribute of entity ***E*** with appropriate measuring units.

For example:

- $LU_i^{(prep\_cost)}$ :: preparation cost associated with land unit *i*
- $P_{i,p}^{(area)}$ :: area of parcel *p* of land unit *i*
- $T_{i,p,t}^{(y,EtOH)}$ :: quantity of ethanol per hectare produced in year *y*, with treatment *t* on parcel *p* of land unit *i*

The following is a list of notations used in the formulas below:

N_{LU} = number of land units

PD = project duration [year]

LU_i = land unit i

$LU_i^{(\max\_area)}$ = maximum area available for LU_i [ha]

$LU_i^{(area)}$ = area of LU_i [ha]

$LU_i^{(prep\_cost)}$ = preparation cost per ha for land unit i [$/ha]

$LU_i^{(rent\_cost)}$ = rent cost per ha for land unit i [$/ha]

$LU_i^{(np)}$ = number of parcels in land unit i

$P_{i,p}$ = parcel p of land unit i

$P_{i,p}^{(area)}$ = area of $P_{i,p}$ [ha]

$P_{i,p}^{(nt)}$ = number of treatments in $P_{i,p}$

$T_{i,p,t}$= treatment t of $P_{i,p}$

$T_{i,p,t}^{(start\_year)}$ = start year of $T_{i,p,t}$

$T_{i,p,t}^{(end\_year)}$ = end year of $T_{i,p,t}$

$T_{i,p,t}^{(prod\_cost)}$ = production cost per ha for treatment $T_{i,p,t}$ [$/ha]

$T_{i,p,t}^{(transp\_cost)}$ = transportation cost per ha for treatment $T_{i,p,t}$ [$/ha]

$T_{i,p,t}^{(y,EtOH)}$ = quantity of ethanol per ha output in year y by treatment $T_{i,p,t}$.

This is expressed in the respective units of the biofuel product under analysis; for the purposes of this chapter and the test case where the biofuel product is ethanol, the volume unit considered in this chapter is litres of ethanol [l/ha]

$EtOH_{min}^{(y)}$= minimum volume of ethanol to be produced in year y [l]

$EtOH_{max}^{(y)}$= maximum volume of ethanol to be produced in year y [l]

$\$(c, y)$ = present value of money of cost c per hectare in year y [$/ha]

$PREP$ = site preparation cost component of the objective function [$]

$RENT$ = land rent cost component of the objective function [$]

$PROD$ = biomass production cost component of the objective function [$]

$TRAN$ = biomass transportation cost component of the objective function [$]

The only decision variables used in the model are $P_{i,p}^{(area)}$.

Objective function

Minimize Z

where,

$$Z = PREP + RENT + PROD + TRAN \tag{2.1}$$

$$PREP = \sum_{i=1}^{N_{LU}} LU_i^{(area)} LU_i^{(prep\_cost)} \tag{2.2}$$

$$RENT = \sum_{i=1}^{N_{LU}} LU_i^{(area)} \sum_{y=0}^{PD-1} \$\left(LU_i^{(rent\_cost)}, y\right) \tag{2.3}$$

$$PROD = \sum_{i=1}^{N_{LU}} \sum_{p=1}^{LU_i^{(np)}} \sum_{t=1}^{P_{i,p}^{(nt)}} P_{i,p}^{(area)} \sum_{y=T_{i,p,t}^{(start\_year)}}^{T_{i,p,t}^{(end\_year)}} \$\left(T_{i,p,t}^{(prod\_cost)}, y\right) \tag{2.4}$$

$$TRAN = \sum_{i=1}^{N_{LU}} \sum_{p=1}^{LU_i^{(np)}} \sum_{t=1}^{P_{i,p}^{(nt)}} P_{i,p}^{(area)} \sum_{y=T_{i,p,t}^{(start\_year)}}^{T_{i,p,t}^{(end\_year)}} \$\left(T_{i,p,t}^{(transp\_cost)}, y\right) \tag{2.5}$$

Subject to:

Area constraints

Area accounting for land unit i:

$$LU_i^{(area)} = \sum_{p=1}^{LU_i^{(np)}} P_{i,p}^{(area)} \tag{2.6}$$

Area availability for land unit i:

$$LU_i^{(area)} \leq LU_i^{(\max\_area)} \tag{2.7}$$

Production constraints

Biofuel product volume output in year y:

$$EtOH_{min}^{(y)} \leq EtOH^{(y)} \leq EtOH_{max}^{(y)} \qquad 2.8$$

$$EtOH^{(y)} = \sum_{i=1}^{N_{LU}} \sum_{p=1}^{LU_i^{(np)}} \sum_{t=1}^{P_{i,p}^{(nt)}} P_{i,p}^{(area)} * \sum_{y\prime=T_{i,p,t}^{(start\_year)}}^{T_{i,p,t}^{(end\_year)}} \delta_{y,y\prime} T_{i,p,t}^{(y\prime,EtOH)} \qquad 2.9$$

where $\delta_{i,j} = \begin{cases} 1 \; if \; i = j \\ 0 \; otherwise \end{cases}$

$$EtOH = \sum_{i=1}^{N_{LU}} \sum_{p=1}^{LU_i^{(np)}} \sum_{t=1}^{P_{i,p}^{(nt)}} P_{i,p}^{(area)} * \sum_{y=T_{i,p,t}^{(start\_year)}}^{T_{i,p,t}^{(end\_year)}} T_{i,p,t}^{(y,EtOH)} \qquad 2.10$$

Non-negativity is assumed on all variables.

2.3.1.1 Objective function description

The objective function (eq. 2.1) is a linear cost minimization equation. It minimizes the overall costs of all biomass production activities: land preparation costs, land rent costs, production costs, and transportation costs. The land preparation and rent costs (eqs. 2.2 and 2.3), as well as number of hectares, are site specific and need to be specified individually for each land unit LU_i under analysis. Following the Model I formulation of Johnson and Scheurman (1977), in each land unit LU_i the optimization algorithm assigns a sequence of regeneration-harvesting treatments $T_{i,p,t}$ to smaller land management units $P_{i,p}$ (further called land parcels) (eq. 2.4). The algorithm determines the area of the land parcels and the optimum treatment sequence $P_{i,p}^{(area)}$.

Each land unit type can have its own list of possible treatments, depending on the specific requirements of the land base, production methods, or biomass resource availability. For example, if a land unit is located on a site with lower productivity soil and no irrigation, then the type of treatments that can be applied to the land unit will correspond to the site conditions. There is no need for an explicit constraint on the time sequence of treatments, because the C-BOS model imposes an implicit built-in constraint on the sequence of treatments (see Figure B.1 for an example).

The biomass production costs (eq. 2.4) are both parcel specific and treatment specific. All biomass production costs can be specified and are calculated on a per hectare, per treatment, per activity, per year basis. Which production costs are included depends on the specific detail required of the analysis. Finally, the biomass transportation costs (eq. 2.5) are calculated similarly to the production costs, and they are considered separately for the ability to track them in the final results.

2.3.1.2 Area constraints

The sum of land parcel areas $P_{i,p}^{(area)}$ belonging to a specific land unit LU_i must add up to the number of hectares of that land unit (eq. 2.6). The maximum area of each land unit is specified in eq. 2.7. The optimization algorithm determines the area of each land parcel, which, as mentioned above, remains constant throughout the planning horizon (*i.e.* the land parcels are not aggregated over time); this is a direct consequence of the model type I implementation technique (Johnson and Scheurman 1977).

2.3.1.3 Production constraints

The biomass production model formulation in this study considers that a certain volume of ethanol needs to be produced each year by the biorefinery, within a specified range (eqs. 2.8 and 2.9). The necessary volume of biomass feedstock (*i.e.* the portion of the harvested biomass that is suitable for conversion and that will actually be converted into ethanol) is calculated directly from the volume of ethanol needed, using a specified biomass conversion efficiency. The volume of ethanol output can be specified and is calculated on a per hectare, per treatment, per activity, per year basis. There are no ending inventory constraints in the model.

2.3.2 Test case description

As a practical application of the proposed carbon-aware biomass optimization planning system C-BOS, we present a test case for a biofuel project producing ethanol in a biorefinery using biomass from a plantation with fast growing poplar trees. The test case represents an illustration of a biomass production system that is configured and designed with typical conditions of the Pacific Northwest. The data used for the test case was sourced from existing commercial or research operations where available, or by compiling best available estimates from published research.

In this section we detail the key assumptions in our analysis for the following elements: land units, transportation of biomass to biorefinery, biomass production strategies (treatments), biomass growth and yield, biomass production activities and costs, biorefinery techno-economic assumptions, ethanol conversion factor, project planning horizon, computer implementation of the C-BOS model, and test case scenarios.

Land units

To demonstrate the model structure, we considered six land unit types for the test case (N_{LU}=6). The land unit types have three key attributes that differentiate them: the capability of the land to be either irrigated or non-irrigated, the land productivity (or soil capability), and the transportation distance to the biorefinery. Three of the land types were non-irrigated (LU_1, LU_3, and LU_5) and three were irrigated (LU_2, LU_4, and LU_6). For the non-irrigated types, LU_1 had high productivity, LU_3 medium productivity, and LU_5 low productivity. Similarly for the irrigated types, LU_2 had a high productivity, LU_4 medium productivity, and LU_6 low productivity (see Table 2.1). For the test case, the medium and low productivities were assumed to be 10% and 30% lower than the high productivity, respectively. These assumptions were found to be reasonable estimates based on discussions with experts on poplar research or commercial activities in the Pacific Northwest (Carlson 2011, Carson 2009, Eaton 2010). However, the model structure allows for the differences in land productivity to be included as decision variables. This will be further explored in a future publication.

In the C-BOS model it is possible to specify the area available for planting $LU_i^{(\max\_area)}$ for each type of land unit. For the test case the available area for each land unit was assumed to be sufficiently large as to not constrain the model.

The site preparation cost was assumed to be 200 \$/ha. The land rent cost was assumed to be 198 \$/ha/yr for the land units closest to the biorefinery (*i.e.* LU_1 and LU_2), 178 \$/ha/yr for LU_3 and LU_4 (*i.e.* 10% lower), and 138 \$/ha/yr for LU_5 and LU_6 (*i.e.* 30% lower). These estimates were found to be reasonable in the context of the potential willingness to pay for private land owners (Eaton 2010).

Transportation of biomass to biorefinery

The transportation distances between land units and biorefinery can be specified in the C-BOS model for each land unit type separately. For the test case the transportation distance was assumed to be low (= 40 km), medium (= 100 km), or high (= 200 km), within the typical distances assumed in the literature (National Research Council 2009).

The transportation of harvested material from land units to the biorefinery assume the use of logging trucks (for transporting whole or bucked stems, which will be further chipped at the biorefinery) and chip trucks (for transporting wood chips from land units at harvest time). The harvested stems from the single stem production treatments (non-irrigated S, and irrigated Si) and from the non-irrigated coppice (C) were assumed to be transported by log truck. The wood chips from the irrigated coppice (Ci) treatments and the harvested branches from the other treatments were assumed to be transported by chip trucks.

The costs of transporting the harvested biomass (logs or chips, depending on the production method of each land parcel) to the biorefinery were calculated on a per-unit and per-kilometre basis using costing parameters referenced in the published literature (McKechnie *et al.* 2011, Zhang *et al.* 2010, Gautam *et al.* 2010, Sambo 2002, Sessions 2010).

For a transportation distance of 40 km, the roads were assumed to be: dirt 2 km, gravel 5 km, and paved 33 km, with respective i) average travel speeds of 4.8 km/h, 24.1 km/h and 88.5 km/h; and ii) average fuel consumption rates of 60 l/h, 45 l/h, and 30 l/h (Sessions 2010). The average payload was assumed to be 18.75 t (dry tonnes), and the average hourly rate was $127/h. The average transport unit cost for logs using log trucks was calculated to be 0.74 $/t/km. Similarly, the average transport unit cost for logs using log trucks was calculated to be 0.36 $/t/km for a transportation distance of 120 km, and 0.29 $/t/km for 200 km.

In this chapter, “t” and “tonne” mean metric tonne. Unless otherwise specified, the weights are expressed on an oven-dry basis.

The transport cost for chips using chips trucks was calculated using an hourly rate of transportation of \$85/h and a load weight of 15.88 t/load (Gautam *et al.* 2010). For a 40 km transportation distance the transport cost for chip trucks was calculated to be 0.52 \$/t/km. Similarly, the transport cost for chips using chips trucks was 0.26 \$/t/km for a transportation distance of 120 km, and 0.21 \$/t/km for 200 km.

The reference cost for diesel fuel used in the transportation cost analysis was assumed to be 1.20 \$/l.

Biomass production strategies (treatments)

For the test case we considered six types of biomass production strategies (treatment types) for fast-growing poplar plantations. The biomass growth and yield data for each treatment are based on existing data from poplar plantation operations and test plots in the Pacific Northwest (Eaton 2010, Carson 2009, Carlson 2011). As the biomass growth and yield data are proprietary, the detailed measurements are not reported in this chapter; instead, the mean annual increment (MAI) of the above-ground biomass (stem, bark and branches) and the rotation length were used to describe the biomass growth and yield. The six biomass production options (treatments) considered in the test case were as follows:

1. Single stem planting of cuttings on non-irrigated land (treatment further called “S”): the high productivity land types were assumed to yield an MAI of 13.35 t/ha/year (stem, bark, and branches) over a 8-year rotation, at a stand density of 1347 trees per hectare; the MAI for medium productivity was assumed to be 10% less than the high MAI (12.01 t/ha/yr), and MAI for low productivity was 30% less than the high MAI (9.34 t/ha/yr).
2. Single stem planting of cuttings on irrigated land (treatment “Si”): MAI for high productivity 17.17 t/ha/year, 6-year rotation, at 1537 trees/ha; the medium MAI was 10% less (15.45 t/ha/yr) and the low MAI was 30% less (12.02 t/ha/year).
3. Short rotation coppice on non-irrigated land (this is a modified coppice system where one dominant shoot is selected and let to grow from the root system, in each rotation period; treatment “C”): the high productivity MAI was assumed

13.35 t/ha/year, over a 15-year rotation (5 years x 3) at 1794 trees/ha; medium MAI was 12.01 t/ha/yr, and low MAI was 9.34 t/ha/yr.

4. Short rotation coppice on irrigated land (treatment “Ci”): MAI for high productivity 17.17 t/ha/year, over a 15-year rotation (3 years x 5) at 3588 trees/ha; medium MAI was 15.45 t/ha/yr, and low MAI was 12.02 t/ha/yr.
5. No-activity (treatment “N”) with duration of 1 year: introduced to allow flexibility in the model in the establishment phase (i.e. in the first seven years) to not plant hectares in a particular land unit starting necessarily with year zero if not needed, since (1) the biorefinery is not active (hence not demanding biomass) until year 8, and (2) the aim was to not sell biomass to the open market but instead use it only within the biofuel project.
6. Idle (treatment “I”): introduced to allow flexibility in the model towards the end of the planning horizon to not plant and grow any additional biomass if not needed; it is expected that this situation arises when the biomass yield and/or the ethanol conversion efficiency increase towards the end of the planning horizon as a result of future technological improvements, thereby reducing the number of hectares needed to produce the necessary quantity of biomass; the treatment duration was chosen to be sufficiently large as needed (i.e. 50 years); to explore the impact of this treatment type on the analysis, in some scenarios this treatment was purposely not made available.

On the irrigated land units the possible treatment types were single stem irrigated (Si) and coppice irrigated (Ci). For the non-irrigated land unit, the treatment types available were single stem non-irrigated (S) and coppice non-irrigated (C). The choice of these treatments was based on biomass production options currently considered in commercial poplar plantations (Eaton 2010), and were assumed to be reasonable options for other geographical regions within the Pacific Northwest. The formulation of the C-BOS model allows for any other combination of land unit and treatment types.

Table 2.1. Test case data by land unit types for treatment, productivity, transport distance, and land rent

			Land Unit					
			LU_1	LU_2	LU_3	LU_4	LU_5	LU_6
Treatment Type	Single Stem Non-irrig.	S	X		X		X	
	Single Stem Irrigated	Si		X		X		X
	Coppice Non-irrig.	C	X		X		X	
	Coppice Irrigated	Ci		X		X		X
Productivity [t/ha/year]*	Non-irrigated	Low					9.34	
		Med			12.01			
		High	13.35					
	Irrigated	Low						12.02
		Med				15.45		
		High		17.17				
Transport distance [km]		Low					40	40
		Med			120	120		
		High	200	200				
Land rent cost [$/ha/year]			198	198	178	178	138	138

* Productivity is expressed as tonnes of biomass (stem, bark, and branch) per hectare per year

Biomass growth and yield

The live biomass growth and yield dynamics over time were represented in the C-BOS model by monitoring six types of live biomass pools: (1) stem, (2) bark, (3) branches, (4) foliage, (5) coarse roots, and (6) fine roots. The biomass MAI values in Table 2.1 represent the total above-ground biomass less foliage (*i.e.* stem, bark, and branch). As C-BOS monitors the biomass growth dynamics in an annual time step the annual change in each biomass pool was referenced to the MAIs.

The stem, bark, branch and foliage biomass components were calculated from the respective MAI values, separately for i) the single stem irrigated and non-irrigated, and the coppice non-irrigated treatments, and ii) the coppice irrigated treatment, as described below.

For the single stem irrigated and non-irrigated, and the coppice non-irrigated treatments, the biomass components are calculated as a proportion of the total above-ground biomass, for each year of the respective rotations and for each production strategy, using the biomass equations for black cottonwood described in Standish *et al.* (1985), as shown in Table of APPENDIX A. The Standish *et al.* equations use tree diameter at breast height (dbh) and height as input parameters. The dbh and height biomass equations were derived through polynomial regressions of 2nd order (R^2=0.994 for dbh, and R^2=0.983 for height respectively) from experimental data of *P. trichocarpa* growth trials in British Columbia (Carson 2009).

The coarse and fine roots biomass is calculated in the C-BOS model as proportion of total above-ground biomass by using the equations from Kurz *et al.* (1996) and Li *et al.* (2003), as referenced in Kurz *et al.* (2009). These are the same equations used in the CBM-CFS3 model for hardwood species, which were assumed to be applicable to the single stem poplar trees analyzed here. The root biomass dynamics as a proportion of the total above-ground biomass are shown in Table in APPENDIX A.

For the coppice, irrigated treatment, since the coppiced shoots have small diameters resembling branches rather than stems, it was assumed that all above-ground biomass would accumulate only in branch and foliage. Just to be clear, stem growth is indeed occurring, however in C-BOS is captured in the branch biomass pool.

For both coppice treatments, the growth of biomass between successive coppicing harvests (within the same 15-year rotation cycle) was assumed to slightly increase from one rotation to the next. This was based on reports from recent experimental tests (Eaton 2010, Berguson 2010) based on which the biomass yield was assumed in this study. For the non-irrigated coppice treatment the biomass yield was assumed to increase by 4% between the first and the second rotation, and by 8% between the first and the third rotation. For the irrigated coppice treatment, the yield was assumed to increase by 2%, 3%, 4% and 5% between the first and respectively the 2nd, 3rd, 4th and 5th rotation. This assumption was considered to be a reasonable estimate (Eaton 2010, Berguson 2010).

Biomass production activities and costs

As mentioned earlier, the quantity of biomass that is necessary to be converted into ethanol is assumed in this chapter to be produced entirely by the biofuel project (*i.e.* no biomass is purchased from, or sold to, the open market). In practical applications it would be conceivable that biofuel projects could purchase some of the necessary biomass from the open market to help with the start-up of the biomass production operations, and to sell excess biomass if too much is produced. However, this could bring additional uncertainties into the modeling framework since we are concerned with the ability to account accurately and consistently for the carbon sequestrations and emissions. For this reason, no biomass sales or purchases to and from the market are considered in this chapter, so that subsequent (*i.e.* discussed in the following chapters) life cycle analyses only include biomass that belongs within the system boundaries of the biofuel project.

The general categories for biomass production activities considered for the test case were: site preparation, planting, silviculture, harvest and processing. These activities were specific to each of the four treatments applied and were referenced from both published and unpublished sources (McAuliffe 2011, Eaton 2010, Gautam *et al.* 2010, Berguson 2010, Timberline Natural Resource Group Ltd. 2009). The costs by treatment for biomass production activities on a per hectare, per year basis, are shown in Table A.3 through Table A.5 in APPENDIX A.

The planting and silviculture activities for the single stem non-irrigated (S) treatment included: labour planting cuttings, herbicide previous land use, ripping, pre-emergent herbicide, pre-plant herbicide, backpack spray (year 1); cultivation, herbicide weed control (years 1-3); fertilization (years 1, 5); management, mapping, GIS, other operational expenses – road maintenance, fire insurance (all years); restoration (year 8). As the production cost data are proprietary, aggregated costs are reported in this chapter instead of detailed costs.

The planting and silviculture activities for the single stem irrigated (Si) treatment included: labour for planting cuttings, drip hose rollout, ripping, pre-emergent herbicide, pre-plant herbicide, backpack spray (year 1); herbicide weed control (years 1, 2); fertilization (years 1, 2, 4); drip hose maintenance and roll-up (years 2, 4, 6); irrigation – capital, management, power, maintenance (years 1, 2, 3-6); pest control, management, mapping, GIS, other operational expenses – road maintenance, fire insurance (all years); restoration (year 6).

The planting and silviculture activities for the coppice non-irrigated (C) treatment (for the total 15 years, *i.e.* 3 rotations of 5 years each) included: labour for planting rods, herbicide for previous land use, ripping, pre-emergent herbicide, backpack spray, (year 1); pre-plant herbicide (years 1, 6, 11); cultivation (years 1, 2-3, 6-7, 11-12); herbicide weed control, years 1-4, 6-8, 11-13); fertilization (years 1, 5, 10); singling, pruning (years 5, 10);

management, mapping, GIS, other operational expenses – road maintenance, fire insurance (all years); restoration (year 15).

The planting and silviculture activities for the coppice irrigated (Ci) treatment included (for the total 15 years, *i.e.* 5 rotations of 3 years each): labour planting cuttings, drip hose rollout, ripping, pre-emergent herbicide, backpack spray (year 1); pre-plant herbicide (years 1, 6, 11); herbicide weed control (years 1-2, 4, 7, 10, 13); drip hose maintenance and roll-up (years 1, 3-4, 6-7, 9-10, 12-13, 15); irrigation – capital, management, power, maintenance (years 1-3, 5-6, 8-9, 11-12, 14-15); fertilization, pest control, management, mapping, GIS, other operational expenses – road maintenance, fire insurance (all years); restoration, year 15.

The harvesting and processing activities differed between the single-stem and the coppice treatments. For the single-stem treatments (S, Si) and the non-irrigated coppice treatment (C) the activities included: i) felling, skidding, de-limbing and loading for stem+bark+branch, ii) mobile chipping of branch biomass at the plantation site, and iii) debarking-chipping of stem+bark biomass at biorefinery.

The costs of harvesting and processing for single-stem treatments were assumed to be: $13.23/green tonne for felling, skidding, de-limbing and loading (applied to stem+bark+branch harvested biomass pools), $3.16/green tonne for debarking and chipping (applied to stem+bark harvested biomass pools; using an electric debarker and chipper at the biorefinery) (Martin et al. 2000), and $4.29/green tonne for mobile grinding (applied to harvested branch biomass) (Gautam et al. 2010). We assumed that 5% of the harvested biomass is not recovered during harvest operations due to mechanical losses (*i.e.* only 95% of the available biomass in harvest years is recovered and transported to the biorefinery).

For the conversion from green tonne to dry tonne we used a coefficient of 0.5 (*i.e.* 50% moisture content on a mass basis). This compares with 48% MC for poplar chips used for biochemical conversion (National Research Council 2009) and as high as 57% MC for irrigated coppice (Stanton 2013).

For poplar wood we used a wood specific gravity of 0.35 (Penman *et al.* 2003).

For the coppice, irrigated treatment (Ci) the harvesting activities were performed with a modified harvester, blower and trailer tractor. The cost of harvesting and processing for irrigated coppice treatments was assumed to be $7/green tonne (Buchholz and Volk 2011).

Although the costs for biomass harvesting and processing shown in Table A.4 and Table A.5 are expressed in $/ha/yr, they were actually implemented in the model as $/t of wood. This offers the flexibility in the model to allow for these costs to change on a per hectare basis as the biomass yield changes, e.g. to increase if the biomass yield (expressed in t wood/ha) increases, as in the test case scenarios.

Biorefinery techno-economic assumptions

For the test case we assumed that the poplar wood chips are converted to ethanol in a biorefinery using an enzymatic hydrolysis and fermentation technology. As demonstrated by recent industrial applications (Larsen *et al.* 2012) this technology is close to commercial scale deployment. The cellulosic-ethanol manufacturing process and the techno-economic assumptions that we considered for the test case are those described in the National Research Council (2009) report using poplar wood chips as feedstock. The National Research Council (2009) design of the cellulosic biorefinery uses a hot water pre-treatment method and includes a lignin-based burner and boiler for the generation of steam and a steam turbine for the generation of electricity.

For the test case we considered a biorefinery annual production capacity $EtOH_{min}$ of 227,124,707 l EtOH, corresponding to 60 million US gallons, the midpoint of the 20-100 mil gallons/year range considered in the National Research Council (2009) report. The upper bound (maximum annual volume) $EtOH_{max}$ was set at 283,905,884 l EtOH, representing 125% of $EtOH_{min}$. It was assumed that the biorefinery had the technical capability of producing the $EtOH_{max}$ volume if needed. It is expected that this situation may arise due to improvements in biomass yield and conversion efficiency towards throughout the project planning horizon, for the case when the idle treatment “I” is not available and therefore the project must use all the hectares available to produce biomass (*i.e.* must use one of the four biomass production treatments), which in turn will increase the volume of ethanol produced beyond the minimum required.

Ethanol conversion factor

The conversion efficiency for white chips (obtained from the debarked stem and the branch biomass in the S, Si, and C treatments) was assumed to be 267.86 l EtOH per tonne of dry wood, calculated from 78 gallons per ton of dry wood (National Research Council 2009); ton represents a US short ton. This value is the midpoint of the 234-299 l/t (calculated from 68-87 gal/ton) range scenarios referenced in the National Research Council (2009) study. The low value of 234 l/t represents little (low) improvements in technology and process efficiency in a biorefinery, and 299 l/t represents major (high) improvements. As a reference point, the National Research Council study, citing (U.S. Department of Energy 2010) reports a best-case theoretical ethanol yield of 442 l/t (106 gal/ton), considering a 100% efficiency of the conversion process. The composition of poplar wood chips assumed in the National Research Council (2009) study (p. 129 Table 3.1 of that study) was: cellulose 40.3%, hemicellulose 22.0%, lignin 23.7%, ash 0.6%, other components 13.4%.

For whole-tree chips (obtained from the stem+bark+branch biomass in the Ci treatment) the conversion factor was assumed to be 240 l EtOH per tonne of dry wood, representing 90% of the yield for white chips, calculated for: Hybrid Poplar DN-34, whole tree chips yield = 97.1 gal/BDT; Hybrid Poplar DN-34, white chips yield = 108.3 gal/BDT (U.S. Department of Energy 2009, U.S. National Renewable Energy Laboratory 2004).

Project planning horizon

The project planning horizon PD was assumed to be 68 years for the test case. At the beginning of year 1 the land units selected for land-use change were assumed to have been prepared and planted. Next, a period of 8 years was set to permit the planted biomass to reach harvesting age (since the maximum rotation age of the production treatments, *i.e.* for treatment S, was 8 years), and to build the biorefinery. The next six decades represent two economic lifetimes for the biorefinery; the lifetime is reported or assumed in the literature to be around 30 years (Aden *et al.* 2002). For the test case it was assumed that after 30 years of service either another biorefinery will be constructed or the existing one will be updated/retrofitted to run for another 30 years, bringing the planning horizon to a total of 68 years. The choice of two economic lifetimes for the biorefinery is within the ranges considered in the published literature (Aden *et al.* 2002).

Computer implementation of the C-BOS model

The linear programming formulation of the C-BOS model was implemented using the GNU MathProg modeling language and solved using the python bindings for the GLPK solver (Makhorin 2011). The input data are extracted from spreadsheets and are processed by a custom python package that formulates the linear programming problem sent for solving to the GLPK engine. Tabular outputs such as the ones shown in APPENDIX B are produced in a web browser using a javascript file created by the package. Output is also created as spreadsheet files suitable for processing by the next stage.

2.3.3 Experimental scenarios for C-BOS model

The linear programming methodology used in the C-BOS model allows the investigation of various sensitivity analyses questions. Two important areas of scientific research in biofuels development are to improve the biomass yield and the conversion efficiency. The C-BOS modeling framework enables the investigation of the effects for both types of improvements.

Also, since the yield improvements are expected to result in less and less plantation area being used to produce the necessary biomass (we recall that the ethanol quantity necessary to be produced annually was constant), in the C-BOS model it would be interesting to investigate the effect of "idle" treatments, which will allow plantation hectares to become idle at some point in time when it is not necessary to use them any longer. The idle treatments are expected to reduce the overall project costs by not taking on planting and maintenance activities on land parcels that will not be harvested by the end of the project.

To demonstrate the practical applicability of the C-BOS model under these potential conditions, we examined eight scenarios, described below, which account for i) the possibility of potential improvements in biomass yield and conversion efficiency throughout the planning horizon, and ii) the availability of the idle treatment "I".

The presence or absence of improvements was considered in order to illustrate the impact of various improvement scenarios on the model results, namely on the objective function (ethanol total production costs), unit cost of ethanol (*i.e.* cost per litre as well as per tonne of wood harvested), and plantation area required (*i.e.* net average area).

In order to analyze the effects of improving either the biomass yield only, or the conversion efficiency only, or both, four different potential situations were considered:

1) no improvements in biomass yield or conversion efficiency over the planning horizon;
2) improvements in biomass yield of 5% per decade[1] (the proportion improvement is calculated from the previous decade);
3) improvements in conversion efficiency of 5% per decade; and,
4) improvements in both biomass yield and conversion efficiency of 5% per decade.

The availability of the idle treatment "I", as well as its absence, was considered in order to explore the impact of two potential production strategies as they may arise in practical applications of biofuel projects, with respect to the hectares used for the biofuel project. One strategy is to continue to use the same number of plantation hectares regardless of whether they are needed for biomass production throughout the project planning horizon or not. The other strategy is to use only as many hectares as needed, *i.e.* use less hectares if improvements in biomass yield and/or conversion efficiency occur. Two potential situations were considered:

A) Idle treatments allowed: in the case when the biofuel project needs to use fewer hectares because of continuous improvements in biomass yield and conversion efficiency, while the minimum ethanol volume needed remains constant throughout the planning horizon reasonable. These scenarios are further referred to as "idle treatment scenarios." The availability of idle treatments is based on the assumption that at the end of the planning horizon all land reverts to another use – at least not in production for ethanol.
B) No idle treatments: in the case when the biofuel project uses a constant number of hectares for biomass production throughout the planning horizon, regardless of the presence of improvements, even if this means producing more than the minimum

[1] This rate was considered to be a reasonable estimate (Stanton 2013)

required volume of biomass (hence more ethanol volume) in later decades of the project, because more hectares are available than are needed. These scenarios are referred further as "non-idle treatment scenarios."

By combining the improvement situations 1-4 and the idle situations A-B, eight scenarios result as shown in Table 2.2.

Table 2.2. Scenario matrix

	Idle treatments allowed	No idle treatments
No improvements in biomass yield or conversion efficiency	Scenario 1A	Scenario 1B
Improvements in biomass yield of 5% per decade	Scenario 2A	Scenario 2B
Improvements in conversion efficiency of 5% per decade	Scenario 3A	Scenario 3B
Improvements in both biomass yield and conversion efficiency of 5% per decade	Scenario 4A	Scenario 4B

The values for the first decade (*i.e.* year 0 to year 9) for biomass yield and conversion efficiency are those described above. The yield values are applied to treatments based on which decade represents the starting year of the respective treatment.

The choice of idle treatments in biomass production strategies poses an interesting problem to the carbon and greenhouse gas balance analyses. If the carbon-GHG balance of the biofuel project is to be calculated for the entire duration of the project horizon, then the question arises about how to model the carbon in the land parcels that have the idle treatment applied to them; *i.e.* what happens with the biogenic carbon in soil organic matter pools on the respective land parcels since no more biomass is purposely grown on those land parcels after the idle treatment is applied? This aspect is beyond the scope of this chapter, and can be an interesting future research aspect on this subject.

2.4 Results and discussion

The results from all eight test case scenarios, as summarized in Table 2.3, were similar to what was anticipated. As expected, the least expensive ethanol unit cost result was in Scenario 4A (where future improvements are assumed in both biomass yield and conversion efficiency, and where idle treatments were allowed), which also utilized the least amount of hectares.

The total project cost (objective function value) was lower in all improvement scenarios with idle treatments (scenarios 2A to 4A) compared with the no-improvement scenario 1A, as expected, since the biomass yield and/or the conversion efficiency were increasingly higher. As anticipated, scenario 4A had the lowest total project cost from all the idle treatment scenarios, due to improvements in both biomass yield and conversion efficiency and use of less hectares.

The total project cost for the improvement scenarios with no idle treatments were lower in scenarios 3B and 4B compared with the no-improvement scenario 1B, while in scenario 2B the total cost was slightly higher; this is explained by the higher harvested biomass volume and the higher volume of ethanol produced in scenario 2B as a consequence of the necessity to use all the plantation hectares. While the model has the objective to minimize overall costs, this is subject to the constraints that are set. In the case of improvement scenarios 2B through 4B, the area constraints, along with the fact that the idle treatments were not available, "forced" the model to use all available hectares and overproduce biomass and ethanol in quantities that were more than the minimum needed (*i.e.* less hectares were needed because of the gradual improvements in biomass yield and conversion efficiency, but the model was forced to use all hectares anyway). The lower bound of the ethanol volume constraint was 13,854,607,127 l (227,124,707 l/year * 61 years), while in scenarios 2B through 4B over 14 million litres were produced.

The model results shown in Table 2.3 demonstrate that project activities were optimized and costs were minimized, as expected. The amount of project activities undertaken in the non-idle treatment scenarios (scenarios B) was higher than the idle treatment scenarios (scenarios A): the volume of ethanol and of biomass harvested was higher in the non-idle treatment scenarios than the correspondent idle treatment scenarios, with the exception of scenarios 1A and 1B (where the volume of ethanol produced and of biomass harvested was the same, as expected, but scenario 1A had lower total costs because it was able to use the idle treatments and not have to undertake activities if not needed).

On a volumetric basis ($/l ethanol), all the cost components of the objective function (*i.e.* site preparation, land rent, biomass production, and biomass transport costs) for the improvement scenarios (all scenarios 2, 3 and 4) were lower than the no-improvement scenarios (both scenarios 1), as expected (Figure 2.1). The harvested wood unit cost on a volumetric basis was progressively lower in the improvement scenarios for both the idle treatment and no idle treatment cases.

The harvested wood unit cost on a mass basis ($/t) was calculated by dividing the total project costs by the total quantity of above-ground biomass harvested, *i.e.* stem, branch and bark. The harvested wood unit cost ($/t) did not exhibit the same continuous reduction for all the improvement scenarios, *i.e.* it was higher in scenario 3A *vs*. scenario 1A, and in 3B *vs*. 1B (Figure 2.2). This is explained partly by the model objective, which was to minimize overall costs and not unit costs, and partly by the fact that the decrease in harvested biomass from scenario 1 to scenario 3 was higher than the decrease in total project costs. The harvested biomass in scenario 3A was 52,637,789 t, which was 11.0% less than the 59,167,192 t of scenario 1A, while the project costs were 10.9% less ($7,326,735,811 *vs*. $8,221,813,564). Similarly, the harvested biomass was 8.5% less in scenario 3B compared with scenario 1B, while the project costs were 6.4% less.

Also, it is important to mention that some of the activities costs are calculated on a per hectare basis, instead of a per tonne of wood basis. As mentioned previously, the silviculture activities costs, as well as the site preparation and the land rent costs, are calculated on a per hectare basis and they are assumed constant regardless of the biomass yield, whereas the harvesting and processing costs are calculated on a per tonne of wood basis and therefore are proportional with the quantify of wood processed.

Table 2.3. C-BOS model results for test case

		Scenario 1A	Scenario 2A	Scenario 3A	Scenario 4A	Scenario 1B	Scenario 2B	Scenario 3B	Scenario 4B
Obj. function value	[$]	8,221,813,564	7,868,821,527	7,326,735,811	7,033,437,915	8,440,966,919	8,460,319,970	7,904,502,011	8,219,793,848
Site prep. cost	[$]	17,807,824	17,530,481	17,359,749	17,276,225	17,807,824	15,865,181	15,862,310	15,220,398
Land rent cost	[$]	858,959,881	758,089,681	762,354,890	675,937,259	908,244,465	813,192,442	813,203,866	783,493,583
Biomass prod. cost	[$]	4,807,316,248	4,574,618,121	4,293,621,709	4,119,550,058	4,977,185,020	5,022,924,902	4,753,483,673	4,891,403,821
Biomass transp. cost	[$]	2,537,729,610	2,518,583,244	2,253,399,463	2,220,674,373	2,537,729,610	2,608,337,446	2,321,952,162	2,529,676,046
Maximum annual area	[ha/yr]	89,039	87,652	86,799	86,381	89,039	79,326	79,312	76,102
Average annual area	[ha/yr]	79,873	71,079	70,997	63,947	84,456	75,606	75,593	72,635
Ethanol produced	[l]	13,854,607,127	13,854,607,127	13,854,607,127	13,854,607,127	13,854,607,127	14,122,149,702	14,082,855,804	15,411,263,085
Site prep. unit cost	[$/l]	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001
Land rent unit cost	[$/l]	0.062	0.055	0.055	0.049	0.066	0.058	0.058	0.051
Biomass prod. unit cost	[$/l]	0.347	0.330	0.310	0.297	0.359	0.356	0.338	0.317
Biomass transp. unit cost	[$/l]	0.183	0.182	0.163	0.160	0.183	0.185	0.165	0.164
Total harvested wood unit cost	[$/l]	0.593	0.568	0.529	0.508	0.609	0.599	0.561	0.533
Conversion cost at biorefinery	[$/l]	0.304	0.304	0.304	0.304	0.304	0.304	0.304	0.304
Ethanol unit production cost	[$/l]	0.897	0.871	0.832	0.811	0.913	0.903	0.865	0.837
Harvested biomass (stem+bark +branch)	[t]	59,167,192	59,233,140	52,637,789	52,733,204	59,167,192	60,833,470	54,148,370	58,898,837
Site prep. unit cost	[$/t]	0.30	0.30	0.33	0.33	0.30	0.26	0.29	0.26
Land rent unit cost	[$/t]	14.52	12.80	14.48	12.82	15.35	13.37	15.02	13.30
Biomass production unit cost	[$/t]	81.25	77.23	81.57	78.12	84.12	82.57	87.79	83.05
Biomass transport cost	[$/t]	42.89	42.52	42.81	42.11	42.89	42.88	42.88	42.95
Total harvested wood unit cost	[$/t]	138.96	132.84	139.19	133.38	142.66	139.07	145.98	139.56

"Maximum annual area" represents the maximum number of hectares of land that were used in any of the planning horizon years, and "average annual area" is the ratio between the sum of hectares used in all years and the number of years.

The area needed for the biofuel project decreased progressively from scenario 1A through 4A, as well as from scenario 1B through scenario 4B. This was expected due to the effect of the improvements in biomass yield and conversion efficiency, requiring less biomass, hence less hectares. For the idle treatment scenarios (1A-4A) the decrease in the net average area used (calculated as the sum of non-idle hectares used each year, divided by the number of project years) was even more substantial, demonstrating that the model results were able to clearly represent the impact of improvements in both biomass yield and conversion efficiency. The average annual area used is shown in detail by land unit type for each of the idle treatment scenarios in Table D.1 through Table D.4.

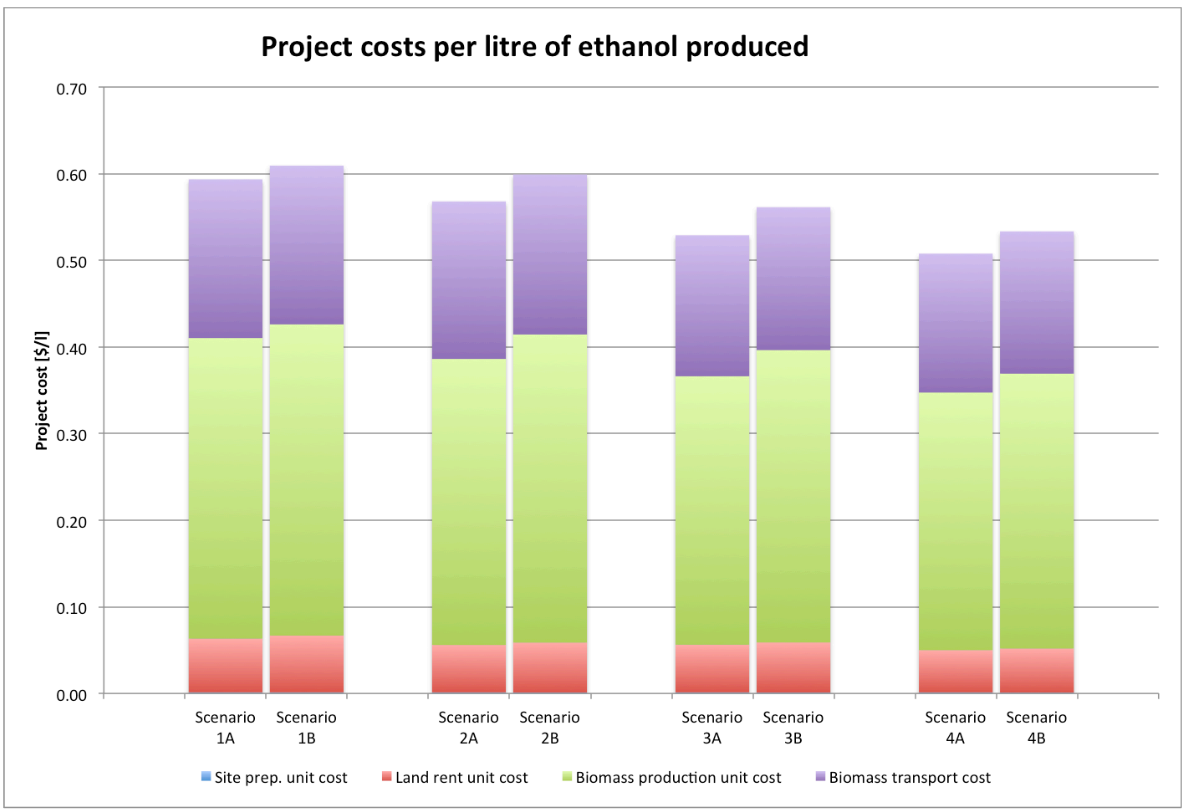

Figure 2.1. Project costs components per litre of ethanol

(Note: Site preparation unit cost values are very small, thus not showing well in the graphs)

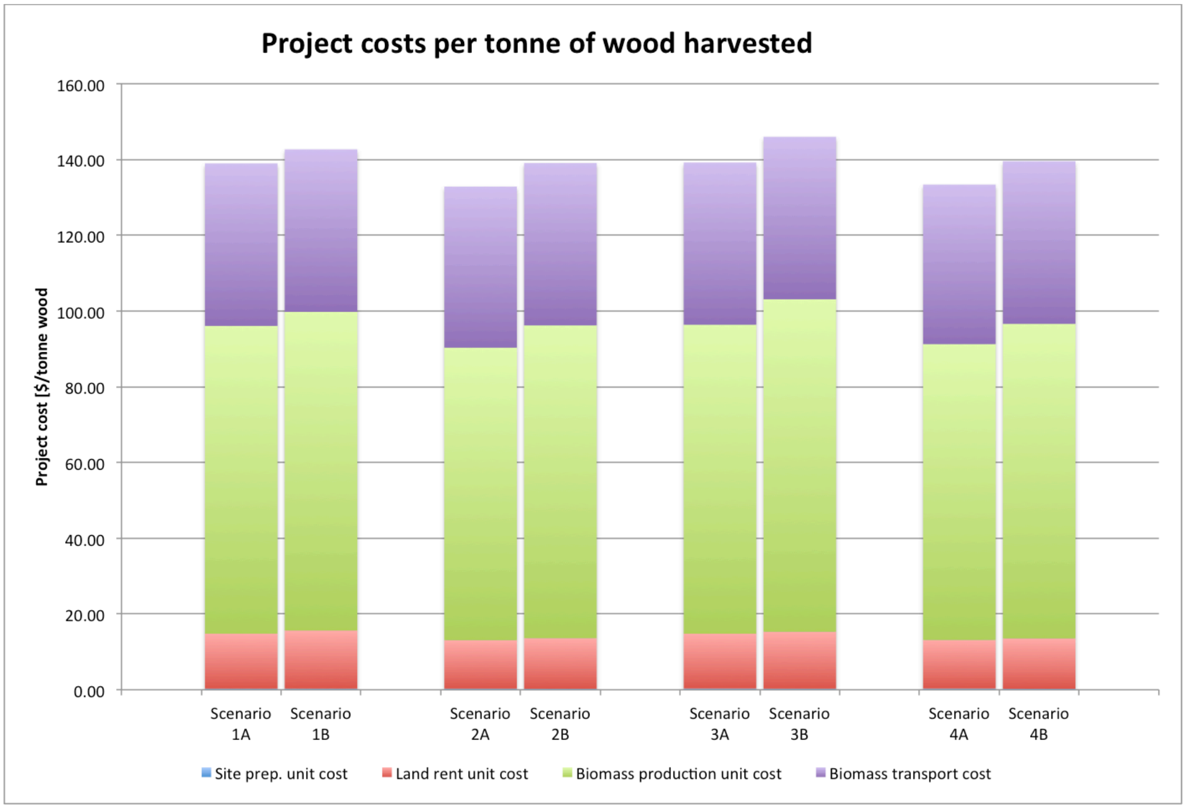

Figure 2.2. Project costs components per tonne of harvested wood

As described in section 2.3.1.1 the solution of the LP optimization model determines how many hectares in each land parcel and with what sequence of harvest-regeneration treatments are used. Each of the land parcels belongs to one of the six land unit types considered (LU_1 through LU_6). One of the advantages of the Type I modeling formulation type used in this study is that the hectares that form any land parcel are preserved intact throughout the entire planning horizon. The quantity of biomass that is grown and harvested in each of the land parcels is accounted from the sequence of harvest-regeneration treatments and from the growth and yield tables, on an annual basis. This is an important capability of the model formulation, which facilitates subsequent determination of the exact quantities in each live biomass pool (*i.e.* stem, bark, branch, foliage, coarse roots, and fine roots), although this is beyond the scope of this chapter.

The detailed results of how many hectares in each land parcel and with what sequence of harvest-regeneration treatments are used are not shown here for all scenarios due to the complexity and amount of the data (up to tens or hundreds of land parcels over 68 years, depending on the scenario). However, in APPENDIX C the cumulative carbon dynamics in all the six live biomass pools (equivalent to the actual biomass quantities) are shown for each scenario.

Detailed model results for scenarios 1A-4A are shown in Table D.1 through Table D.4, and for scenarios 1B-4B in Table D.5 through Table D.8, in APPENDIX D.

Since not all hectares are used in all years, the average annual area used is always less than the total area used. In all scenarios A and B there are fewer hectares used at the beginning of the project (first 8 years) because the biomass production is gradually ramping up, and in scenarios 1A to 4A there are also less hectares used towards the end of the project due to the option to use the idle treatments.

It is interesting to note that the model did not choose to use mainly the land units that produced the highest biomass yield (LU_1 and LU_2), nor those that were within the closest transportation distance (LU_5 and LU_6). This is explained by the trade-off between the low cost of using high yield biomass (LU_1 and LU_2) and the low cost of using land units within a short transportation distance (LU_5 and LU_6). It is not unexpected that the model chose to employ most of the hectares within land unit LU_3, which had a medium biomass yield and located within a medium transportation distance.

The biomass transport cost is an important component of the total biomass cost, as shown in Table 2.3 and Figure 2.1, and this may be a key reason why the land units farther from the biorefinery (*i.e.* land units 1 and 2) were not part of the solutions in a substantial way, even though those were the most productive lands (*i.e.* with the highest biomass yields).

Figure 2.3 and Figure 2.6 show the average annual area used in each of the six land units in scenario 1A through 4A. As expected, the figures illustrate that fewer and fewer hectares are used towards the end of the planning horizon, as greater improvements in both biomass yield and conversion efficiency are considered from scenario 1A to 4A.

Scenario 1A uses 89,039 hectares (all of them located within LU_3) until 7 years before the end of the planning horizon (year 61), after which the number of hectares employed decreases sharply. This can be explained by the fact that, starting with year 61, the model applies the idle treatment to an increasing number of hectares, since there is no incentive to incur the costs of re-planting the hectares that will not have a chance to be harvested by the end of the project (there are no ending inventory constraints in the model). Towards the end of the planning horizon, the model abandons more and more hectares and depletes the biomass stock. In the last year of the project, only about 10,000 hectares are used, all being harvested in that year.

The same pattern of decreasing hectares towards the end of the planning horizon is present in scenarios 2A, 3A and 4A (Figure 2.4 through Figure 2.6). However, there are two distinct "phases" of the hectares reduction in LU_3. In the first phase, between years 32-61 in scenarios 2A and 4A and years 26-61 in scenario 3A, the number of hectares used is decreasing due to the improvements in biomass yield and conversion efficiency. In the second phase (after year 61) the number of hectares used is decreasing due to yield improvements and also to abandoning plantations because planning horizon is near (*i.e.* just enough biomass is grown to produce ethanol until year 68).

The hectares used in the no-idle treatment scenarios (1B to 4B) are not shown graphically in this section, since the number of hectares used is constant throughout each of the respective scenarios (except in the first 8 years, when the number of hectares used is gradually increasing).

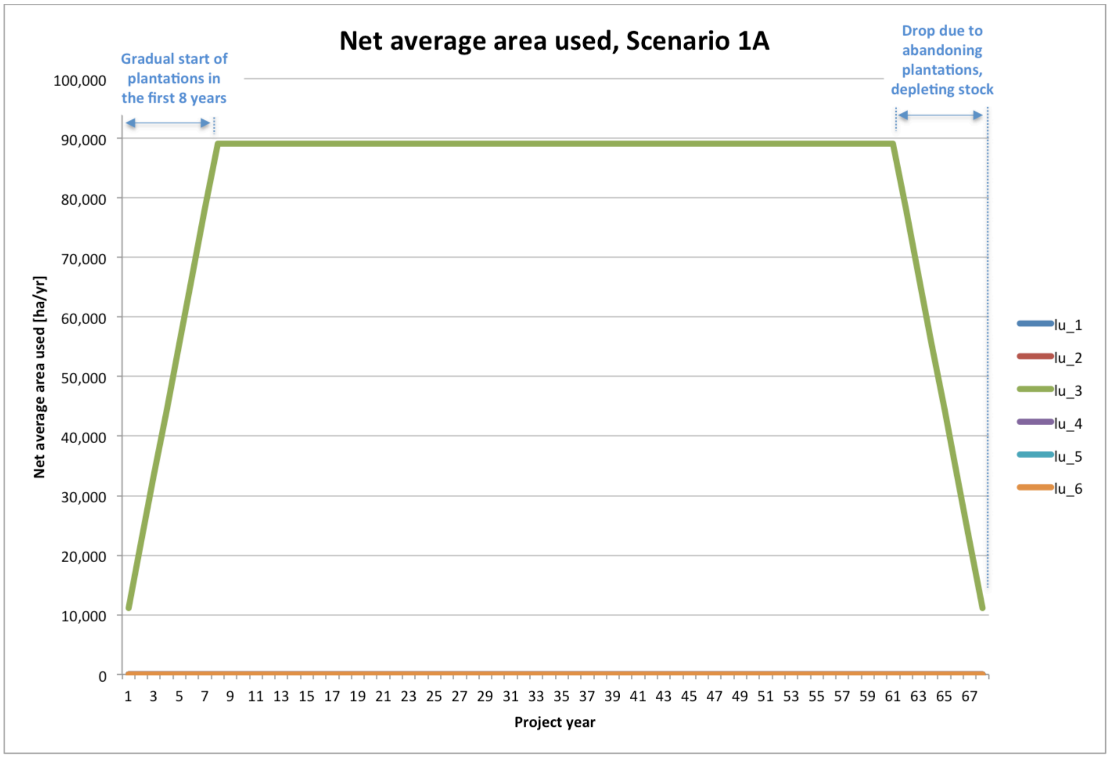

Figure 2.3. Net average area used, scenario 1A

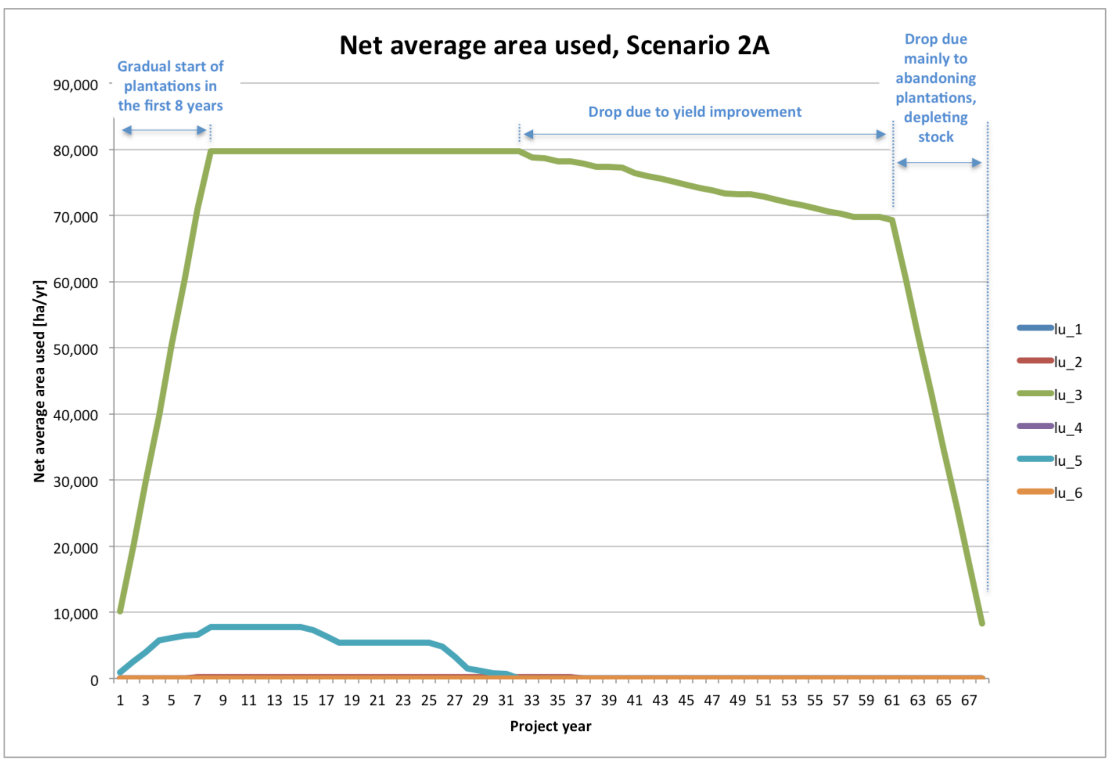

Figure 2.4. Net average area used, scenario 2A

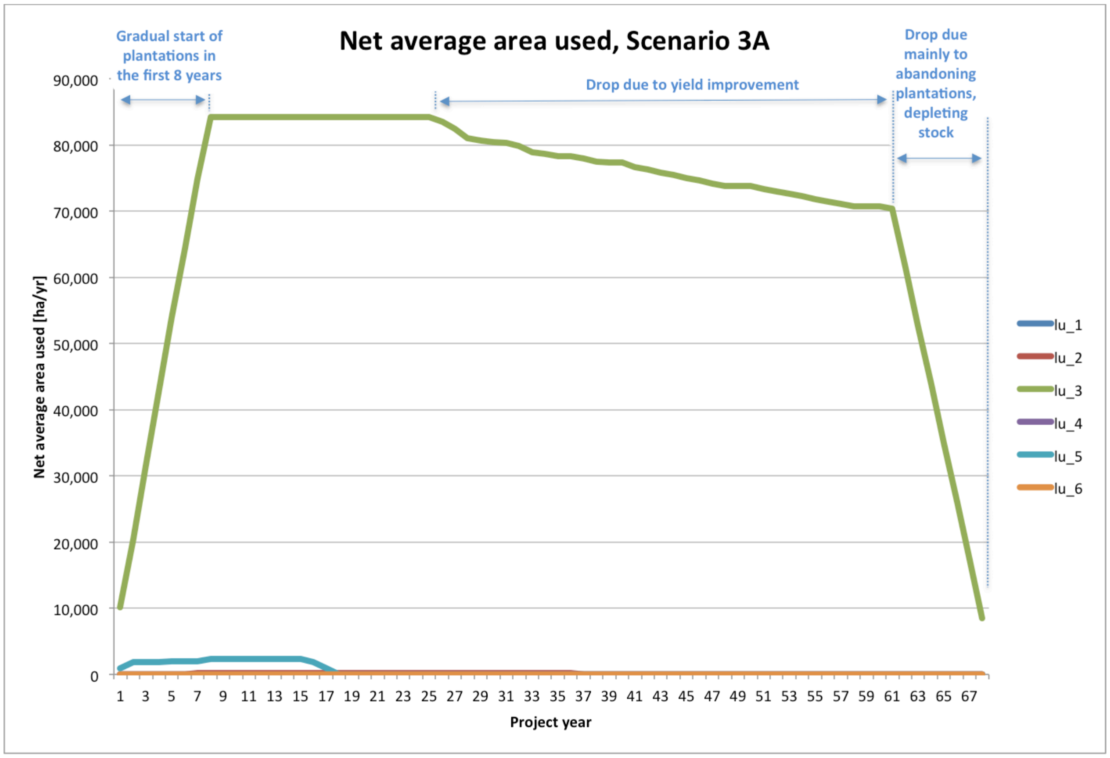

Figure 2.5. Net average area used, scenario 3A

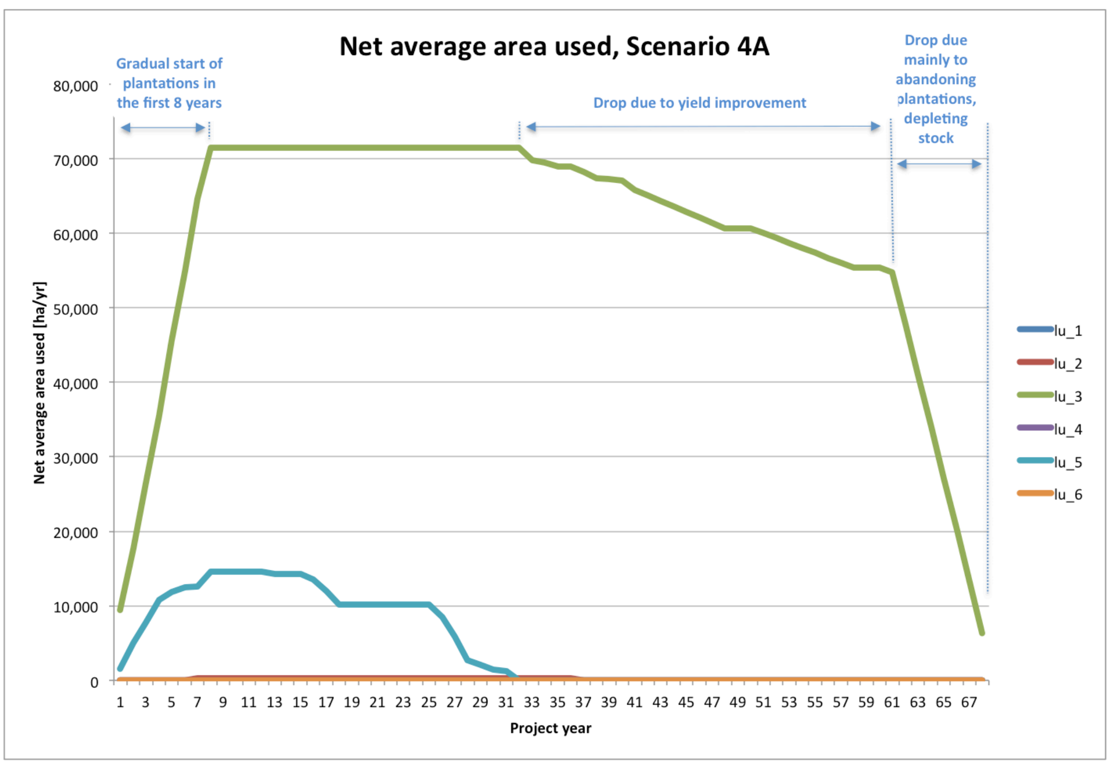

Figure 2.6. Net average area used, scenario 4A

2.4.1 Limitations of the model

A disadvantage of the Type I model formulation, from the perspective of computational requirements, is due to the potentially large size of the LP problem. The number of model variables can be very large because the number of possible regeneration-harvesting sequences for all the land parcels increases as the planning horizon expands, *i.e.* the decision tree shown in APPENDIX B becomes very large very quickly as more years and the respective possible regeneration-harvesting sequences are added to the planning horizon. For scenarios A (idle treatments allowed) the model matrix had 11,730 variables, 67 constraints, and 11,724 possible treatment sequences (*i.e.* possible land parcels). The model run time total for scenarios A was 43.44 seconds, of which LP obj. function solve time was 2.94 sec. For scenarios B (no idle treatments) the model had 5,922 variables, 67 constraints, and 5,916 possible treatment sequences. The model run time total for scenarios B was 20.38 seconds, LP obj. function solve time 0.92 sec.

Another limitation of the model formulation, inherent to linear programming techniques in general, is that it is deterministic. That is, all the possible treatments and their associated growth and yield tables, production costs, must be specified *a priori*. For example, uncertainty is not considered specifically in the model formulation. We have addressed this by assuming that the growth and yield values already adequately capture the inherent risk of reduced yields due to various unfavourable conditions.

The linearity of the objective function can also be considered a limitation of the model. For example, it is assumed that the improvements in biomass yield and in conversion efficiency result in proportional increases or decreases in the cost of various production activities that are dependent on the respective yields. In the absence of more robust published data, we consider this assumption to be satisfactory.

2.5 Conclusion

This chapter proposes the <u>C</u>arbon-aware <u>B</u>iomass production <u>O</u>ptimization <u>S</u>ystem (C-BOS), which models a biomass production system in such a way that all the carbon-related impacts (*i.e.* sequestration and emissions) on biomass stocks and on the landscape can be subsequently quantified over a time horizon. The main outcomes of the C-BOS model are 1) an optimal tactical plan for scheduling regeneration-harvesting biomass production activities for land parcels within land units over the project time horizon, and 2) a detailed account of biomass dynamics over time by individual biomass pools within each land parcel planning unit.

To our knowledge this is the first implementation of a linear programming harvest scheduling model Type I for a life cycle analysis of biofuel production systems, in such a way that individual parcels of land are not aggregated with other land parcels over the planning horizon. This enables the subsequent monitoring of biomass (carbon) dynamics for biogenic carbon pools within each land parcel, both live biomass pools and DOM pools. In terms of live biomass, C-BOS monitors not only the harvested biomass, but all the biomass pools (stem, bark, branch, foliage, coarse roots, fine roots). The explicit monitoring of individual parcels and their respective biomass pools through the planning horizon allows for the determination of biogenic carbon sequestration during tree growth, as well as the biogenic carbon removal at harvest time. This is key for determining the biogenic carbon balance of the biofuel project by directly accounting for the carbon transfers between atmosphere and the organic matter produced by the biofuel project.

The utility of this model was shown on a test case with eight scenarios for short rotation poplar production in the Pacific Northwest, by considering potential future gains in biomass growth yield and conversion to ethanol efficiency, biomass production options (treatments) with different rotation ages, different tree species, transportation distances, and land productivity types, as well as accounting for potential future improvements in biomass growth yield and/or conversion efficiency. The C-BOS model produced results similar to what was expected, namely lower production costs, as well as lower number of hectares needed, for the improvement scenarios compared with the non-improvement scenarios.

The C-BOS model is designed for subsequent integration with a carbon accounting model and a life-cycle biomass-to-biofuel GHG balance model, as decision-support tools for assessing the viability of displacing gasoline with ethanol from wood as a climate mitigation strategy. The carbon accounting model will be described in CHAPTER 3 and the life cycle GHG balance model will be presented in CHAPTER 4.

CHAPTER 3 Modeling the life-cycle biogenic carbon balance of afforestation-to-biofuel projects with the Biogenic Carbon Dynamics model (Bio-CarbD)

3.1 Synopsis

This chapter introduces the life cycle Biogenic Carbon balance and Dynamics model (Bio-CarbD), which simulates the life cycle biogenic carbon balance of a biofuel project, and the dynamics of biogenic carbon pools by individual land planning units throughout the planning horizon of the project. The Bio-CarbD model is based on a mass balance methodology for quantitative analysis of the life cycle biogenic carbon transfer and dynamics between the atmosphere, live biomass pools, and dead organic matter pools, at the project level. Bio-CarbD was developed with the capability to be integrated with the carbon-aware biomass planning optimization system C-BOS introduced in CHAPTER 2.

A novel approach of the Bio-CarbD model is that it does not make any assumptions about the "carbon-neutrality" of the portion of the biomass generated by the biofuel project that is reconstituted in the biofuel product. Instead, Bio-CarbD considers all carbon stocks and accounts for all carbon removals from the atmosphere through natural growth processes of above- and below-ground biomass, carbon transfers between live biomass pools and dead organic matter pools, carbon emissions from biodegradation of organic material through both DOM decay and biomass to biofuel conversion processes, as well as biogenic carbon emissions from the final combustion of biomass-derived biofuels.

In addition, Bio-CarbD also monitors the biogenic carbon pools of individual land planning units, hence it was designed to be integrated with C-BOS, a biomass planting-harvesting model Type I that keeps land planning contiguous over time.

The Bio-CarbD model is a key component of the Carbon Balance and Biomass to Biofuel Optimization planning system (C3BO) described later in CHAPTER 4, and it is further used in the research work of CHAPTER 5.

3.2 Introduction

From an organic matter perspective, the biomass production site (plantation forest) is both a sink of biogenic[2] carbon through photosynthesis and sequestration in live biomass and dead organic matter, and a source of GHG emissions from decay of dead organic matter. In a forest ecosystem, carbon is stored in areas referred to as carbon pools, which include above-ground biomass, below-ground biomass, soil, and litter. The total carbon content of the carbon pools is referred to as the "carbon stock."

Despite significant efforts to develop robust methods, there is currently no consensus on how to account for temporary removals of carbon from, or additions to, the atmosphere in life cycle assessment and carbon footprint accounting (Brandao *et al.* 2013). Carbon neutrality is not necessarily true for all biofuel production systems, and uncertainties and a lack of data make biofuel production C fluxes difficult to accurately quantify (Bonin and Lal 2012). Biomass is often considered to be a carbon neutral feedstock, but a significant amount of GHG emissions are released during the life cycle. For example, during fertilizer production and use, during the transportation of the biomass, as well as in the conversion stages (Borrion *et al.* 2012). As such, the assumption of biogenic carbon neutrality is not valid under policy relevant time horizons if carbon stock changes in the forest are not accounted for (Agostini *et al.* 2013). The assumption of biomass carbon neutrality has been challenged and concerns have been directed particularly at biofuels and wood biofuel, as some may provide only marginal emission savings on a life-cycle basis, or even result in increased emissions (Searchinger *et al.* 2008, Johnson 2009, Maness 2009, Searchinger *et al.* 2009, Tilman *et al.* 2009, O'Laughlin 2010, Cherubini *et al.* 2011, McKechnie *et al.* 2011, Frieden *et al.* 2012, Schulze *et al.* 2012, Smith and Searchinger 2012, Wibe 2012, Zanchi *et al.* 2012, Agostini *et al.* 2013).

[2] For the purposes of this study, biogenic or biotic carbon is defined as the carbon contained in biologically based materials other than fossil fuels (Lal 2008, U.S. EPA 2011). Biologically based feedstocks are non-fossilized and biodegradable organic material originating from modern or contemporarily grown plants including products, byproducts, residues, and wastes from agriculture, forestry, and related industries. It does not include materials such as coal, petroleum, natural gas, and products that are ultimately derived from biologic materials but are not renewable on policy-relevant time frames.

The Intergovernmental Panel on Climate Change (IPCC) suggest several methodologies for monitoring forest carbon stocks and stock changes in the Good Practice Guidance (GPG) for Land Use, Land-use Change and Forestry (LULUCF) (Penman *et al.* 2003). One of these methods is the "one inventory plus change" method, meeting the greatest (Tier 3) degree of estimation certainty. This method requires inventories for forest and land, data for land-use changes, forest management activities, natural disturbances, and detailed models for estimating growth rates and decay. In Canada, the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) is an implementation of the Tier 3 IPCC-GPG standard and the "one inventory plus change" method (Kurz *et al.* 2009).

The CBM-CFS3 carbon budgeting methodology was used by Neilson *et al.* (2006, 2007, 2008) to investigate carbon storage in forest and wood products. Similar to the approach of Hennigar *et al.* (2008), the dynamics of biomass and DOM carbon stocks are obtained *a priori* from CBM-CFS3, and then the C yields of merchantable stem biomass (considered as the equivalent of timber yields) were used as input in WoodStock-Remsoft (Jamnick and Walters 1993), which is a model Type II modeling approach to forest growth and management. Hennigar *et al.* (2008) used stand projections of merchantable volumes of timber as input into CBM-CFS3 to generate carbon yields stored in 14 carbon pools, then grouped the stand-level carbon in four carbon pools. The limitations of this approach (*i.e.* using a Model Type II) were recognized: the application of C yield curves in Woodstock is unlikely to fully conserve C (Neilson *et al.* 2006).

In this chapter the CBM-CFS3 carbon budgeting methodology is used as a reference in combination with a different approach to forest growth and management: C-BOS, a model Type I previously introduced in CHAPTER 2.

The Bio-CarbD model introduced in this chapter is a carbon budgeting methodology that considers and monitors not only the harvested stem biomass and a few selected carbon pools as previously mentioned in the literature, but all the 14 live biomass and dead organic matter pools used in CBM-CFS3. Moreover, Bio-CarbD in combination with C-BOS is capable of representing the land parcels and biomass/organic matter pools in such a way that they can be subsequently quantified for a life cycle carbon balance analysis, by quantifying the biomass pools and the respective biogenic carbon pools on land parcels that are kept contiguous through time.

3.3 Methodology

The Bio-CarbD model and methodology developed in this thesis describes both the biogenic carbon dynamics and the biogenic carbon balance of biomass production strategies. For carbon dynamics, Bio-CarbD uses the "one inventory plus change" carbon accounting method suggested by the IPCC-GPG (Penman *et al.* 2003), implemented in a manner similar to the CBM-CFS3 model (Kurz *et al.* 2009), and using a one-year time step. Bio-CarbD employs the biomass turnover factors, litterfall transfer rates, and decay dynamics parameters as described in the CBM-CFS3. For carbon balance, Bio-CarbD uses equation 3.4 described later in this section.

3.3.1 Biogenic carbon pools

Similar to the CBM-CFS3 (Kurz *et al.* 2009), the Bio-CarbD model accounts for the carbon transfers between carbon pools and atmosphere, and among carbon pools. Bio-CarbD monitors the dynamics of 15 different organic matter carbon pools on a yearly basis, and both above-ground and below-ground carbon is tracked in 6 live biomass pools, and 9 dead organic matter (DOM) pools categorized by their specific rate of decay.

Live biomass pools:

- *bm_stem*: live stem biomass
- *bm_bark*: live bark biomass
- *bm_branch*: live branch biomass
- *bm_foliage*: live foliage biomass
- *bm_coarse_roots*: live coarse roots biomass
- *bm_fine_roots*: live fine roots biomass

Dead organic matter pools:

- *dom_sng_stem*: snag stems
- *dom_sng_branch*: snag branches
- *dom_medium*: medium decaying DOM
- *dom_ag_fast*: above-ground fast decaying DOM

- *dom_ag_very_fast*: above-ground very fast decaying DOM
- *dom_ag_slow*: above-ground slow decaying DOM
- *dom_bg_fast*: below-ground fast decaying DOM
- *dom_bg_very_fast*: below-ground very fast decaying DOM
- *dom_bg_slow*: below-ground slow decaying DOM

Table 3.1 shows the correspondence between the carbon pools considered in Bio-CarbD, CBM-CFS3, and those recommended in the IPCC-GPG.

Table 3.1. Correspondence between pools in the Bio-CarbD model, in CBM-CFS3, and in GPG

Bio-CarbD Model pool	**CBM-CFS3 pool**	**Description**	**GPG pool**
bm_stem	Merchantable plus bark	Live stemwood not including bark	Above-ground biomass
bm_bark	Merchantable plus bark	Bark on live stemwood	Above-ground biomass
bm_branch	Other wood plus bark	Live branches, including bark	Above-ground biomass
$bm\_foliage$	Foliage	Live foliage	Above-ground biomass
bm_coarse_roots	Coarse roots	Live roots, approximately ≥5mm diameter	Below-ground biomass
bm_fine_roots	Fine roots	Live roots, approximately <5mm diameter	Below-ground biomass
dom_sng_stem	Snag stems DOM	Dead standing stemwood	Dead wood
dom_sng_branch	Snag branches DOM	Dead branches, stumps and small trees including bark	Dead wood
dom_ag_medium	Above-ground medium DOM	Coarse woody debris on the ground	Dead wood
dom_ag_fast	Above-ground fast DOM	Fine and small woody debris plus dead coarse roots in the forest floor, approximately ≥5 and <75mm diameter	Litter
$dom\_ag\_very\_fast$	Above-ground very fast DOM	The L horizon (Soil Classification Working Group 1998) comprised of foliar litter plus dead fine roots, approximately <5mm diameter	Litter
dom_ag_slow	Above-ground slow DOM	F, H and O horizons (Soil Classification Working Group 1998)	Litter
dom_bg_fast	Below-ground fast DOM	Dead coarse roots in the mineral soil, approximately ≥5 diameter	Dead wood
$dom\_bg\_very\_fast$	Below-ground very fast DOM	Dead fine roots in the mineral soil, approximately <5mm diameter	Soil organic matter
dom_bg_slow	Below-ground slow DOM	Humified organic matter in the mineral soil	Soil organic matter

To account for the harvested biomass, Bio-CarbD includes three harvested biomass pools:

- *hbm_stem*: harvested stem biomass
- *hbm_bark*: harvested bark biomass
- *hbm_branch*: harvested branch biomass

For carbon accounting and carbon balance purposes, Bio-CarbD also includes a carbon pool representing the carbon that is absorbed from the atmosphere through biomass growth, and a carbon pool representing the releases (emissions) to the atmosphere from biomass losses and DOM decay:

- *CO2_absorbed*: carbon absorbed from the atmosphere by photosynthesis of growing biomass
- *CO2_released*: carbon released to the atmosphere through decay of DOM pools by natural processes or as a result of the impact of mechanized harvest-planting activities

Since Bio-CarbD accounts only for the biogenic carbon in live biomass and dead organic matter pools, it does not include emissions from fossil fuels or from energy production activities of the biomass production system. Also, the substitution credits for displacement of gasoline (also known as fuel switch credits) are not considered here. These elements will be the subject next in CHAPTER 4, where a full life cycle carbon and GHG analysis will be described.

3.3.2 Biogenic carbon pools dynamics

The year-to-year dynamics of carbon stocks in all carbon pools is calculated using the "one inventory plus change" method suggested by IPCC-GPG (Penman *et al.* 2003). As mentioned earlier, the Bio-CarbD model tracks the carbon stocks and fluxes at the land parcel level. Land parcels are the planning units whose area and optimal treatment sequence determined by the optimization algorithm, and are subsets of the larger land units. Each land parcel has a set of carbon pools, CP_x, where $x \in \{$*bm_stem, bm_bark, bm_branch, bm_foliage, bm_coarse_roots, bm_fine_roots, dom_sng_stem, dom_sng_branch, dom_ag_medium, dom_ag_fast, dom_ag_very_fast, dom_ag_slow, dom_bg_fast, dom_bg_very_fast, dom_bg_slow, hbm_stem, hbm_bark, hbm_branch, CO2_dom*$\}$. The amount of carbon stored in pool x in project year y is $CP_x(y)$ and it changes in time according to the following evolution equation:

$$CP_x(y+1) = \sum_{p \in pools} CTM_{p \to x}^{(y)} CP_p(y) \tag{3.1}$$

where, $CTM_{p \to x}^{(y)}$ is the carbon transfer matrix element that indicates what fraction of the pool p is converted into pool x in the given year y. $CTM_{p \to x}^{(y)}$ depends on the treatment that is active on the parcel in year y. The total amount of carbon is the sum of the carbon pools over all parcels.

To model the carbon dynamics within each parcel on an annual basis, one of the following 3 types of carbon transfer matrices is applied each year to the carbon pools of each land parcel:

1. For all years with the exception of the harvest years and the initial year of the project: a treatment independent matrix that describes the natural decay of DOM pools, with no influence from mechanized activities (Table 3.2). This matrix is a direct implementation of the CBM-CFS3 biomass turnover and litterfall transfer matrices for live biomass pools, and the decay and physical transfer parameters for the DOM pools.
2. For harvest years: a treatment dependent matrix that accounts for the additional impact of the mechanized activities on the transfer of carbon between pools. Since the impact of mechanized activities is different between treatments (e.g., different harvesting machinery and methods), and it also depends on the magnitude of the impacts, these matrices are dependent on the treatment type and on the assumed impacts, as follows:
    a. A matrix applied in harvest years (example in Table 3.3)
    b. A matrix applied in intermediary harvest years to treatments that have an intermediary harvest stage (example in Table 3.4)

3. For the initial year of the project at land-use change (LUC), when carbon losses are caused by impacts of mechanized activities: a matrix applied only once in the project start year, at land-use change (see an example in Table 3.5)

For the no-activity treatment (N), described earlier in CHAPTER 2, it was assumed that the carbon sources equal the carbon sinks; this characterizes the equilibrium condition before the project. For the idle (I) treatment it was assumed that no biomass is growing on the land units.

Table 3.2. Carbon transfer matrix applied to all treatments, and all years except harvest years.

	bm_ stem	bm_ bark	bm_ branch	bm_ foliage	bm_ coarse_ roots	bm_ fine_ roots	dom_ sng_ stem	dom_ sng_ branch	dom_ mediu m	dom_ ag_ fast	dom_ ag_ very_fast	dom_ ag_slow	dom_ bg_fast	dom_ bg_very_ fast	dom_ bg_slo w	CO2_ dom
bm_stem	**0.9955**						0.0045									
bm_bark		**0.9955**					0.0045									
bm_branch			**0.9700**					0.0075		0.0225						
bm_foliage				**0.0500**							0.9500					
bm_coarse_roots					**0.9800**					0.0100			0.0100			
bm_fine_roots						**0.3590**					0.3205			0.3205		
dom_sng_stem							**0.9493**		0.0320			0.0032				0.0155
dom_sng_branch								**0.8283**		0.1000		0.0122				0.0596
dom_medium									**0.9626**			0.0064				0.0310
dom_ag_fast										**0.8565**		0.0244				0.1191
dom_ag_very_fast											**0.6450**	0.0657				0.2893
dom_ag_slow												**0.9790**			0.0060	0.0150
dom_bg_fast													**0.8565**		0.0244	0.1191
dom_bg_very_fast														**0.5000**	0.0850	0.4150
dom_bg_slow															**0.9967**	0.0033

$CTM_{p \to x}^{(y)}$ from equation 3.1 is element from row p, column x.

In Table 3.2, the coefficients in bold typeface are calculated as *1 – (sum of all other values in row)*. All other coefficients are calculated using the parameters for biomass turnover and litterfall transfer for live biomass pools, and the decay and physical transfer parameters for the DOM pools from Kurz *et al.* (2009).

The matrices in Table 3.2 through Table 3.5 are read by rows from left to right, and can be interpreted as follows: for the *bm_stem* pool: 99.55% of the carbon in the pool stays in the *bm_stem* pool and 0.45% is transferred to the *dom_sng_stem* pool. Similarly, for the *dom_sng_stem* pool, 94.93% of the carbon stays in *dom_sng_stem* pool, 3.20% is transferred to the *dom_medium* pool, 0.32% is transferred to the *dom_ag_slow* pool, and 1.55% is transferred to the *CO2_dom* pool. For the *dom_bg_very_fast* pool, 50% stays in the *dom_bg_very_fast* pool, 8.50% is transferred to the *dom_bg_slow* pool, and 41.50% is transferred to the *CO2_dom* pool.

The *CO2_dom* pool represents the carbon that is “lost” from the terrestrial biogenic carbon pools through natural decay of DOM as either natural decomposition processes or as a result of the impact of mechanized activities (soil compaction and scarification).

Table 3.3. Example of a carbon transfer matrix applied to all treatments only in final harvest years

	dom_ sng_ stem	dom_ sng_ branch	dom_ medium	dom_ ag_ fast	dom_ ag_ very_fast	dom_ ag_slow	dom_ bg_fast	dom_ bg_very_ fast	dom_ bg_slow	hbm_ stem	hbm_ bark	hbm_ branch	CO2_ dom
bm_stem	*0.0545*									**0.9455**			
bm_bark	*0.0545*										**0.9455**		
bm_branch		*0.0575*		0.0225								**0.9200**	
bm_foliage					*1.0000*								
bm_coarse_roots				*0.5000*			*0.5000*						
bm_fine_roots					*0.5000*			*0.5000*					
dom_sng_stem	**0.8993**		**0.0320**			0.0032							*0.0655*
dom_sng_branch		**0.7783**		0.1000		0.0122							*0.1096*
dom_medium			**0.9126**			0.0064							*0.0810*
dom_ag_fast				**0.7815**		0.0244							*0.1941*
dom_ag_very_fast					**0.5450**	0.0657							*0.3893*
dom_ag_slow						**0.9290**			0.0060				*0.0650*
dom_bg_fast							**0.7815**		0.0244				*0.1941*
dom_bg_very_fast								**0.4000**	0.0850				*0.5150*
dom_bg_slow									**0.9467**				*0.0533*

In Table 3.3 the coefficients identified in italics typeface are different from those in the Table 3.2 matrix, as follows:

- *bm_stem*, *bm_bark* and *bm_branch*: 5% of the biomass carbon in these pools is assumed to be transferred to their DOM corresponding pools (e.g., *bm_stem* to *dom_sng_stem*) as a result of harvesting activities. For example, the coefficient of transfer from *bm_stem* to *dom_sng_stem* is 0.0545 = 0.0045 + 0.0500; where 0.0045 is the coefficient of transfer used in non-harvest years.
- *bm_foliage*: it is assumed that during harvest 100% of the foliage is transferred to the *dom_ag_very_fast* pool, since this is the pool where foliage matter is transferred through natural processes. The coefficient of transfer from *bm_foliage* to *dom_ag_very_fast* is 1.0000.
- *bm_coarse_roots* and *bm_fine_roots*: it is assumed that at final harvest the tree roots are not removed (*i.e.* dug out) from the site but are purposely eradicated with herbicide, and therefore their biogenic matter is transferred to the pools where it is naturally transferred, *i.e.* to *dom_ag_fast* and *dom_bg_fast* for coarse roots, and to *dom_ag_very_fast* and *dom_bg_very_fast* for fine roots.

- the coefficients in column *CO2_dom* are calculated as follows: each coefficient is a sum between i) the correspondent coefficient in the natural carbon transfer matrix in Table 3.2 and ii) the carbon loss coefficient assumed for harvest activities (this is discussed in detail in section 3.3.5.5 below). For example, the coefficient for *dom_sng_stem* is 0.0655, which is the sum of 0.0155 (from Table 3.2) and 0.0500 (the assumed coefficient for the loss of soil due to mechanized activities in harvest years, Table 3.10). For *dom_sng_branch* the coefficient is 0.1096 = 0.0596 + 0.0500, and so on for the other coefficients in column *CO2_dom*.

Similar to Table 3.2, the coefficients in bold typeface in Table 3.3 are calculated as *1 – (sum of all other values in row)*, and all other coefficients are calculated using the parameters from Kurz *et al.* (2009).

Table 3.4. Example of a carbon transfer matrix applied to all treatments only in intermediate harvest years

	bm_ coarse_ roots	bm_ fine_ roots	dom_ sng_ stem	dom_ sng_ branch	dom_ medium	dom_ ag_ fast	dom_ ag_ very_fast	dom_ ag_slow	dom_ bg_fast	dom_ bg_very_ fast	dom_ bg_slow	hbm_ stem	hbm_ bark	hbm_ branch	CO2_ dom
bm_stem			0.0545									0.9455			
bm_bark			0.0545										0.9455		
bm_branch				0.0575		0.0225								0.9200	
bm_foliage							1.0000								
bm_coarse_roots	**0.9900**					0.0100									
bm_fine_roots		**0.3590**					0.3205			0.3205					
dom_sng_stem			**0.9243**		0.0320			0.0032							0.0405
dom_sng_branch				**0.8033**		0.1000		0.0122							0.0846
dom_medium					**0.9376**			0.0064							0.0560
dom_ag_fast						**0.8190**		0.0244							0.1566
dom_ag_very_fast							**0.5950**	0.0657							0.3393
dom_ag_slow								**0.9790**			0.0060				0.0150
dom_bg_fast									**0.8565**		0.0244				0.1191
dom_bg_very_fast										**0.5000**	0.0850				0.4150
dom_bg_slow											**0.9967**				0.0033

In Table 3.4 the coefficients in column *CO2_dom* are calculated similar to Table 3.3, with the difference that the impact of mechanized activities for intermediate harvest is assumed to be only 50% compared with that of activities for final harvest. For example, the coefficient for *dom_sng_stem* is 0.0405 = 0.0155 + 0.0500 * 50%. The assumption of reduced impact of intermediate harvest activities compared with final harvest and planting activities is justified by the fact that during intermediate harvests the root system is left intact and also there are no planting preparation activities, such as plowing.

Similar to the previous carbon transfer tables, the coefficients in bold typeface in Table 3.4 are calculated as *1 – (sum of all other values in row)*, and all other coefficients are calculated using the parameters from Kurz *et al.* (2009)

Table 3.5. Example of a carbon transfer matrix applied to all treatments only once at land-use change

	dom_ medium	dom_ ag_ fast	dom_ ag_ very_fast	dom_ ag_slow	dom_ bg_fast	dom_ bg_very_ fast	dom_ bg_slow	CO2_ dom
bm_foliage			1.00					
bm_coarse_roots		0.50			0.50			
bm_fine_roots			0.50			0.50		
dom_sng_stem								
dom_sng_branch								
dom_medium	*0.843958*							0.1560
dom_ag_fast		*0.755895*						0.2441
dom_ag_very_fast			*0.585675*					0.4143
dom_ag_slow				*0.86*				0.1400
dom_bg_fast					*0.755895*			0.2441
dom_bg_very_fast						*0.46*		0.5400
dom_bg_slow							*0.8717*	0.1283

At land-use change it is assumed that the pre-existing above-ground biomass in the *bm_stem*, *bm_bark*, and *bm_branch* pools is harvested or otherwise removed as part of the site preparation activities. This assumption is consistent to current approaches in practice (Hansen et al. 1993, Mitchell et al. 1999, Isebrands 2007, Berguson 2010, Eaton 2010). The removed biomass is included in a separate carbon balance calculation for the biofuel project (see CHAPTER 4), however is not directly monitored in the Bio-CarbD model and therefore not shown in Table 3.5.

The pre-existing above-ground biomass in the *bm_foliage* pool is assumed to be transferred to the *dom_ag_very_fast* pool at land-use change (Table 3.5), so it has a coefficient of transfer of 1.00. The below-ground biomass in coarse roots and fine roots is assumed to be transferred equally to *dom_ag_fast* and *dom_bg_fast*, and respectively to *dom_ag_very_fast* and *dom_bg_very_fast*, similar to Table 3.3.

In Table 3.5 the coefficients in column *CO2_dom* are calculated as follows: each coefficient is a sum between i) the correspondent coefficient in the natural carbon transfer matrix in Table 3.2, and ii) the carbon loss coefficient assumed for land-use change activities (this is discussed in detail in section 3.3.5.4 below). For example, the coefficient for *dom_medium* is 0.1560, which is the sum of 0.0310 (from

Table 3.2) and 0.1250 (the assumed coefficient for the loss of soil carbon due to mechanized activities in harvest years, section 3.3.5.4). For *dom_ag_fast* the coefficient is 0.2441 = 0.1191 + 0.1250, and so on for the other coefficients in column *CO2_dom*.

3.3.3 Biomass production system boundary and carbon fluxes

For the purpose of calculating the biogenic carbon fluxes and balance for the biomass production system (*i.e.* biogenic carbon in the plantation areas, including biomass and soil) in relation to the atmosphere, the system boundary is defined in Figure 3.1.

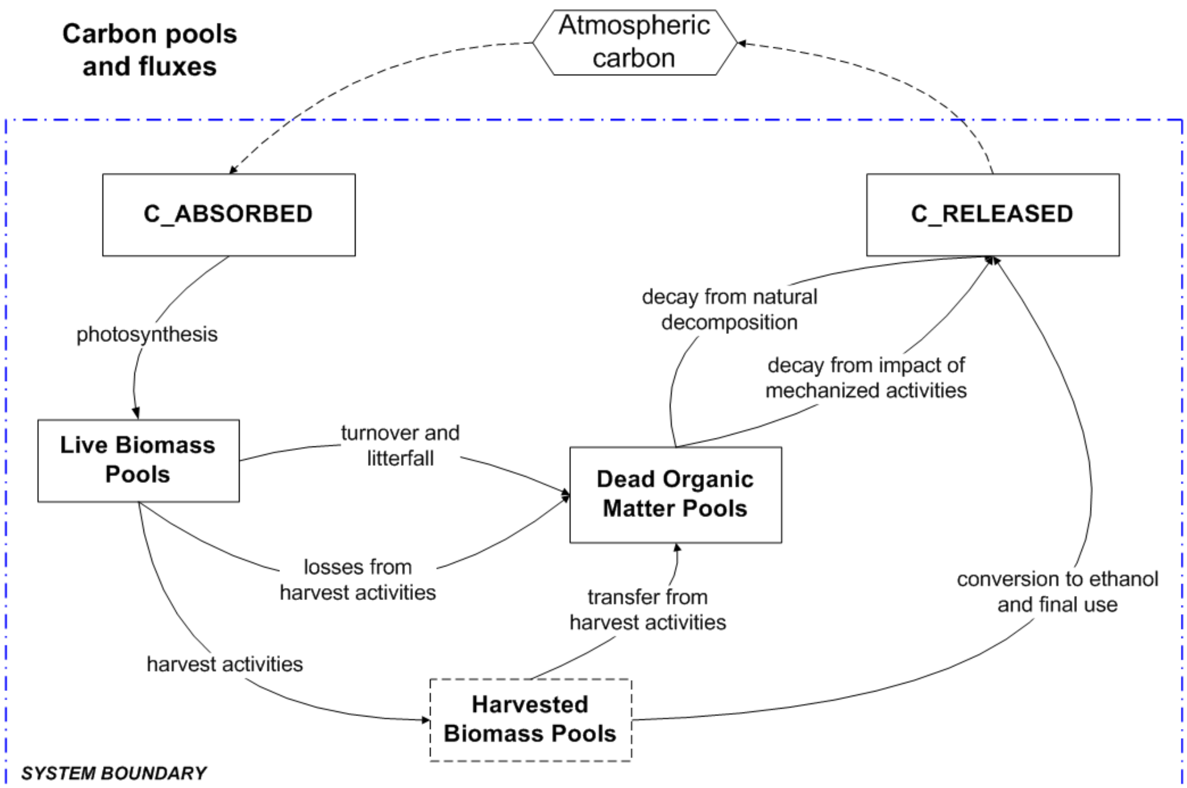

Figure 3.1. System boundary and carbon pools and fluxes.

Similar to the carbon cycle modeling approach described in Kurz *et al.* (2009), from the quantity of carbon dioxide absorbed from the atmosphere by live biomass through processes of photosynthesis, a portion is emitted back into the atmosphere through processes of autotrophic respiration, and the remainder is accumulated in the live biomass pools. In Figure 3.1, the carbon quantity C_ABSORBED represents this net difference between the total CO_2 absorbed from atmosphere by growing biomass and the CO_2 emitted back to atmosphere through respiration. In other words, this study is consistent with the CBM-CFS3 in how it considers the CO_2 absorbed and emitted by living biomass through respirations. Biogenic carbon is transferred between live biomass pools and DOM pools, among DOM pools, and from DOM pools to the atmosphere by a variety of mechanisms: natural decay, physical transfer and turnover of organic matter, and disturbance (*i.e.* harvest-planting mechanized activities). Biomass turnover and litterfall processes cause the transfers of carbon from live biomass to DOM pools. Natural decay processes cause the carbon in DOM to be released to the atmosphere as direct C emissions, and physical transfer processes cause the transfer of C between DOM pools. Carbon that remains in the organic matter pools eventually ends up in the below-ground slow DOM pool. The Bio-CarbD model uses the same yearly turnover rates, litterfall transfer rates, decay parameters, and physical transfer parameters as those reported by Kurz *et al.* (2009) for hardwood species.

The disturbances considered in the Bio-CarbD model are those caused by mechanized site preparation, silviculture and harvesting activities – which represent (1) the loss of biogenic carbon from the ecosystem to the atmosphere through removal of biomass, conversion to biofuel, and final use of biofuel product, and (2) the combination of transfer of biogenic carbon among DOM pools and loss to the atmosphere caused by mechanized land use activities: ripping, plowing, harvesting, initial vegetation removal at land-use change and plantation establishment. The latter carbon losses reflect the increasing number of reports showing that establishing and tending plantations often results in early soil carbon loss (Hansen 1993, Grigal and Berguson 1998, Samson et al. 1999, Paul et al. 2003, Mao et al. 2010, Zhang et al. 2010, Delucchi 2011). The effects on biomass growth and yield of other natural disturbances (e.g., losses or mortality due to insects, pests, or fire) are assumed to be included in the mean annual increment numbers for biomass yield.

The detailed biogenic carbon accounting methodology used in the Bio-CarbD model follows the CBM-CFS3 model and considers the following carbon fluxes, from the perspective of the biofuel project, on a yearly basis (see also Figure 3.1):

- From C_ABSORBED to Live Biomass Pools:
    - During one year, a quantity of atmospheric carbon is absorbed through photosynthesis as CO_2 and accumulated in biomass and organic matter as biomass growth (from C_ABSORBED to "Live Biomass Pools" in Figure

3.1). This represents a transfer of carbon (CO_2 equivalent) from the atmosphere to live biomass pools

  - By the end of the year, some of the absorbed carbon by Live Biomass Pools will remain in live biomass pools (*i.e.* sequestered in live biomass), while some will be transferred to other pools

- From Live Biomass Pools to Dead Organic Matter Pools:
  - Carbon transfer through biomass turnover and litterfall
  - Carbon transfer due to harvest activities (*i.e.* small technical losses during harvest activities)
- From Live Biomass Pools to Harvested Biomass Pools:
  - Carbon transfer by removal of biomass through harvesting activities
- From Dead Organic Matter Pools to C_RELEASED:
  - Carbon emissions from natural decomposition and decay of dead organic matter
  - Biogenic carbon emissions from decomposition and decay caused by the impact on soil of mechanized harvest-planting activities (*i.e.* soil compaction and scarification)
- From Harvested Biomass Pools to Dead Organic Matter Pools:
  - Biogenic carbon transfer from harvested biomass that is not transported to the conversion facility and instead is left on site (*i.e.* for single stem treatments, it is assumed that only the stem and bark are hauled to biorefinery, while the branches and foliage are left on site)
- From Harvested Biomass Pools to C_RELEASED:
  - Biogenic carbon emissions through the processes of conversion of harvested biomass into ethanol, which can be categorized as: 1) the quantity of biomass carbon that is recovered from the conversion process and is used as source of energy input to the process (*i.e.* the recovered lignin used to produce electricity that can power the process or be sold to the grid); 2) the quantity of carbon contained in the ethanol produced and which is released back in the atmosphere during the final use of the ethanol in internal combustion engines; and, 3) the remainder quantity of biomass carbon that

is lost during various stages of processing and does not end up neither in the ethanol produced or in the recovered lignin.

3.3.4 Annual biogenic carbon balance

The carbon balance for a land parcel on a yearly basis can be described as follows: at the end of the analysis year, some of the carbon absorbed from the atmosphere (C_ABSORBED) will remain sequestered in live biomass pools and in DOM pools (C_SEQUESTERED), and the remainder is released back to the atmosphere (C_RELEASED) either through decay and decomposition, or through conversion to ethanol processes and final use of ethanol. The conservation of the natural carbon at the land parcel level is described by equations 3.2-3.4 which are used in the Bio-CarbD model as a self-consistency check.

$$0 = C\_ABSORBED_y + C\_SEQUESTERED_y + C\_RELEASED_y \tag{3.2}$$

$$C\_SEQUESTERED_y = C\_SEQ\_LBIO_y + C\_SEQ\_DOM_y \tag{3.3}$$

$$C\_RELEASED_y = C\_HBIO_y + C\_DCAY_y \tag{3.4}$$

where,

$C\_ABSORBED_y$ = atmospheric carbon (carbon dioxide equivalent) absorbed through photosynthesis by biomass growth during year *y*. This is a negative quantity, by convention in this dissertation. Only part of this carbon ends up in live biomass pools at the end of year *y* ($C\_SEQ\_LBIO_y$), because through natural processes one part of the removed carbon is transferred to DOM pools ($C\_SEQ\_DOM_y$), a second part is harvested and is added to the harvested biomass pools (C_HBIO_y), and a third part is lost back to the atmosphere through decomposition and decay of DOM pools through natural processes or as a result of the impact of mechanized activities (C_DCAY_y). The measuring unit is kilograms of carbon (kg C).

$C\_SEQUESTERED_y$ = carbon that remains sequestered in live biomass pools and in DOM pools at the of year *y* (kg C).

$C\_RELEASED_y$ = carbon that is released to the atmosphere during year *y* through decay of DOM pools, through emissions from all processes of biomass production and conversion to ethanol, and through emissions from the final use of ethanol (kg C).

$C\_SEQ\_LBIO_y$ = carbon that at the end of year *y* remains sequestered in live biomass pools (*bm_stem, bm_bark, bm_branch, bm_foliage, bm_coarse_roots, bm_fine_roots*), after some of the carbon absorbed by live biomass is transferred to DOM pools and some is harvested (kg C).

$C\_SEQ\_DOM_y$ = carbon that at the end of year *y* remains sequestered in dead organic matter pools (*dom_sng_stem, dom_sng_branch, dom_ag_medium, dom_ag_fast, dom_ag_very_fast, dom_ag_slow, dom_bg_fast, dom_bg_very_fast, dom_bg_slow*), after some of the DOM carbon is released through decay (kg C).

C_HBIO_y = the sum of carbon mass in harvested biomass pools: the quantity of carbon at the end of year *y* in harvested biomass pools (*hbm_stem, hbm_bark, hbm_branch*) (kg C).

C_DCAY_y = the sum of carbon mass emitted as CO_2 from decay and decomposition of DOM pools through natural processes and caused by the impact of mechanized activities on soil in harvest years through soil scarification and compaction and subsequent plowing in preparation for planting (more details in section 3.3.5.5 below) (kg C).

The terms in equations 3.2-3.4 are determined as follows:

- $C\_SEQ\_LBIO_y$ and C_HBIO_y are a direct result from the C-BOS model. The C-BOS model determines for each parcel in every year how much biomass (hence, carbon) accumulates in each of the six live biomass pools. $C\_SEQ\_LBIO_y$ represents the sum of these carbon accumulations in all live biomass pools in year *y* for each land parcel. The C-BOS model also determines whether year *y* is a harvest year for the respective parcel and, if yes, how much live biomass is harvested.
- $C\_SEQ\_DOM_y$ and C_DCAY_y are calculated using the biomass growth and yield tables from the C-BOS model and the carbon transfer matrices described in section 3.3.2; C_DCAY_y is calculated using the coefficients in the CO2_dom column.

Since the land parcels and their respective carbon pools are monitored individually through time, this approach ensures that all carbon fluxes are correctly accounted for from one year to the next.

Since the Bio-CarbD model can be configured with many options and inputs, a test case with certain parameters was selected in order to illustrate how the model works and to evaluate several biomass production scenarios.

3.3.5 Test case description

3.3.5.1 Land units, biomass growth and yield, biomass production strategies, project planning horizon

For consistency and clarity, the parameters used for land units, biomass growth and yield, biomass production strategies, and project planning horizon for this test case are the same as those described in section 2.3.2 for the test case of the model C-BOS in CHAPTER 2.

3.3.5.2 Initial carbon stocks in soil including forest floor

The dynamics of soil carbon and dead organic matter is included in the CBM-CFS3 model (Kurz *et al.* 2009), for both above-ground (woody litter and organic soil horizons) and below-ground (mineral soil horizons) components. We follow the CBM-CFS3 approach and include the dynamics of soil carbon and dead organic matter in the Bio-CarbD model.

It was assumed in this study that the initial quantity of the carbon stock in soil was 80 t C/ha, and it was considered that this is a reasonable estimate of the potential carbon content of soil for the test case. In practical applications, however, this quantity will differ depending on the specific conditions of the project. This compares with values assumed in the literature of 5-80 t C/ha in Nave *et al.* (2010); 134-189 t C/ha in Tyner *et al*. (2010) and Searchinger *et al.* (2008); 20-130 t C/ha in IPCC (2006); 71-130 t C/ha in IPCC - Penman *et al.* (2003); 112-117 t C/ha Jobagi and Jackson (2000); 43-226 t C/ha Shaw *et al.* (2005); 90-180 t C/ha Delucchi (2003); 75 t C/ha in GHGenius 3.19a ((S&T)2 Consultants Inc. 2010).

The following section describes the assumptions and calculations used for the distribution of the initial soil carbon among the nine DOM pools.

Initial proportions of carbon in DOM pools

We assume that the soil carbon pools before the start of the biofuel project (*i.e.* prior to land-use change) were at equilibrium, in the sense that the quantity of carbon entering DOM pools from live biomass pools was equal with the quantity of carbon lost through natural processes of DOM decay and decomposition. The equilibrium assumption for the state of the plantation in the absence of the biofuel project (*i.e.* the status quo, or baseline) will be useful later for comparing it with the state of the plantation during the biofuel project. Since the carbon entering DOM pools from live biomass pools is equal with the carbon being emitted through DOM decay and decomposition, the state of equilibrium of the non-project land is maintained throughout the planning horizon (*i.e.* the carbon pools are still at equilibrium at the end of the project). It is acknowledged that this assumption can be viewed as an oversimplification of the actual carbon content and dynamics of the non-project land. However, if the actual carbon dynamics of the landbase in the absence of the project would be known, or assumed to follow a certain development through time, it would be mathematically equivalent to compare this with the biofuel project's dynamics instead.

The proportions of individual biomass pools from the total biomass were calculated using the biomass equations and from Standish *et al.* (1985) for black cottonwood. We described these calculations earlier in CHAPTER 2, section 2.3.2. The resulting proportions, shown in Table 3.6, are similar to the biomass ratios reported by Peichl (2006).

Table 3.6. Proportion of carbon content of live biomass pools from total live biomass carbon

	Live biomass carbon pools						
	M_{bm_stem}	M_{bm_bark}	M_{bm_branch}	$M_{bm\_foliage}$	$M_{bm\_coarse\_roots}$	$M_{bm\_fine\_roots}$	Total Biomass
Proportion from total biomass	47.76%	8.67%	16.79%	4.54%	18.78%	3.46%	100%

To calculate the initial quantities of carbon in DOM pools, we used as a reference the carbon transfer matrix from Kurz *et al.* (2009), which relates the yearly changes in the nine DOM pools to the mass accumulated in the live and DOM pools according to the following equations:

$$\Delta_{dom\_ag\_fast} = 0.0225\ M_{bm\_branch} + 0.01\ M_{bm\_coarse\_roots} - 0.1435\ M_{dom\_ag\_fast} + 0.1\ M_{dom\_sng\_branch} \quad 3.5$$

$$\Delta_{dom\_ag\_slow} = 0.0032\ M_{dom\_sng\_stem} + 0.0122\ M_{dom\_sng\_branch} + 0.0064\ M_{dom\_medium} + 0.0244\ M_{dom\_ag\_fast} - 0.021 M_{dom\_ag\_slow} + 0.0657\ M_{dom\_ag\_very\_fast} \quad 3.6$$

$$\Delta_{dom\_ag\_very\_fast} = 0.95\ M_{bm\_foliage} + 0.3205\ M_{bm\_fine\_roots} - 0.355\ M_{dom\_ag\_very\_fast} \quad 3.7$$

$$\Delta_{dom\_bg\_fast} = 0.01\ M_{bm\_coarse\_roots} - 0.1435\ M_{dom\_bg\_fast} \quad 3.8$$

$$\Delta_{dom\_bg\_slow} = 0.006\ M_{dom\_ag\_slow} + 0.0244\ M_{dom\_bg\_fast} - 0.0033 M_{dom\_bg\_slow} + 0.085\ M_{dom\_bg\_very\_fast} \quad 3.9$$

$$\Delta_{dom\_bg\_very\_fast} = 0.3205\ M_{bm\_fine\_roots} - 0.5\ M_{dom\_bg\_very\_fast} \quad 3.10$$

$$\Delta_{dom\_medium} = 0.032\ M_{dom\_sng\_stem} - 0.0374\ M_{dom\_medium} \quad 3.11$$

$$\Delta_{dom\_sng\_branch} = 0.0075\ M_{bm\_branch} - 0.1718\ M_{dom\_sng\_branch} \quad 3.12$$

$$\Delta_{dom\_sng\_stem} = 0.0045\ M_{bm\_stem} + 0.0045\ M_{bm\_bark} - 0.0507\ M_{dom\_sng\_stem} \quad 3.13$$

At equilibrium we assume that there is no yearly change in any DOM. That is, the quantity of carbon that enters a DOM pool during a year must be equal with the quantity of carbon that exits that pool. We find the equilibrium condition by solving the following system of equations (referencing the equations above):

$$\Delta_{dom\_ag\_fast} = 0 \quad 3.14$$

$$\Delta_{dom\_ag\_slow} = 0 \quad 3.15$$

$$\Delta_{dom\_ag\_very\_fast} = 0 \quad 3.16$$

$$\Delta_{dom\_bg\_fast} = 0 \quad 3.17$$

$$\Delta_{dom\_bg\_slow} = 0 \quad 3.18$$

$$\Delta_{dom\_bg\_very\_fast} = 0 \quad 3.19$$

$$\Delta_{dom\_medium} = 0 \quad 3.20$$

$$\Delta_{dom\_sng\_branch} = 0 \quad 3.21$$

$\Delta_{dom\_sng\_stem} = 0$ 3.22

By solving the above system of equations we find the following relations between the amount of carbon in the DOM and live biomass pools (also summarized in Table 3.7):

$M_{dom\_ag\_fast} = 0.1872\ M_{bm\_branch} + 0.0697\ M_{bm\_coarse\_roots}$ 3.23

$M_{dom\_ag\_slow} = 0.0364\ M_{bm\_stem} + 0.0364\ M_{bm\_bark} + 0.2429\ M_{bm\_branch} + 8.3690\ M_{bm\_foliage} + 0.0810\ M_{bm\_coarse\_roots} + 2.8235\ M_{bm\_fine\_roots}$ 3.24

$M_{dom\_ag\_very\_fast} = 2.6761\ M_{bm\_foliage} + 0.9028\ M_{bm\_fine\_roots}$ 3.25

$M_{dom\_bg\_fast} = 0.0697\ M_{bm\_coarse\_roots}$ 3.26

$M_{dom\_bg\_slow} = 0.06622\ M_{bm\_stem} + 0.0662\ M_{bm\_bark} + 0.4416\ M_{bm\_branch} + 15.2165\ M_{bm\_foliage} + 0.6623\ M_{bm\_coarse\_roots} + 21.6442\ M_{bm\_fine\_roots}$ 3.27

$M_{dom\_bg\_very\_fast} = 0.641\ M_{bm\_fine\_roots}$ 3.28

$M_{dom\_medium} = 0.0759\ M_{bm\_stem} + 0.0759\ M_{bm\_bark}$ 3.29

$M_{dom\_sng\_branch} = 0.0437\ M_{bm\_branch}$ 3.30

$M_{dom\_sng\_stem} = 0.0888\ M_{bm\_stem} + 0.0888\ M_{bm\_bark}$ 3.31

Table 3.7. Equilibrium matrix of carbon accumulation in soil

	M_{bm_stem}	M_{bm_bark}	M_{bm_branch}	$M_{bm\_foliage}$	$M_{bm\_coarse\_roots}$	$M_{bm\_fine\_roots}$
$M_{dom\_ag\_fast}$	0	0	0.1872	0	0.0697	0
$M_{dom\_ag\_slow}$	0.0364	0.0364	0.2429	8.3690	0.0810	2.8235
$M_{dom\_ag\_very\_fast}$	0	0	0	2.6761	0	0.9028
$M_{dom\_bg\_fast}$	0	0	0	0	0.0697	0
$M_{dom\_bg\_slow}$	0.0662	0.0662	0.4416	15.2165	0.6623	21.6442
$M_{dom\_bg\_very\_fast}$	0	0	0	0	0	0.6410
M_{dom_medium}	0.0759	0.0759	0	0	0	0
$M_{dom\_sng\_branch}$	0	0	0.0437	0	0	0
$M_{dom\_sng\_stem}$	0.089	0.089	0	0	0	0

The equilibrium proportion of the DOM pools corresponding to the live biomass proportion from Table 3.6 is shown in Table 3.8.

Table 3.8. Proportion of carbon in DOM pools at equilibrium

	Proportion of total DOM
$M_{dom\_ag\_fast}$	1.74%
$M_{dom\_ag\_slow}$	21.62%
$M_{dom\_ag\_very\_fast}$	5.96%
$M_{dom\_bg\_fast}$	0.51%
$M_{dom\_bg\_slow}$	65.39%
$M_{dom\_bg\_very\_fast}$	0.87%
M_{dom_medium}	1.67%
$M_{dom\_sng\_branch}$	0.29%
$M_{dom\_sng\_stem}$	1.96%
Total	100.00%

As shown in Table 3.8, the below-ground slow decaying DOM pools ($M_{dom\_bg\_slow}$) have the largest initial proportions of carbon, followed by the above-ground slow decaying DOM pools ($M_{dom\_ag\_slow}$).

3.3.5.3 Initial carbon stocks in vegetation

The initial quantity of vegetation (above- and below-ground live biomass existing before the project) was assumed to be correlated to the initial quantity of organic matter in the DOM pools (80 t C/ha), which was at equilibrium as described above. Since the quantity of carbon entering DOM pools from live biomass pools was equal with the quantity of carbon lost through natural processes of DOM decay and decomposition, we needed to calculate that quantity of vegetation that would generate the necessary quantity of matter entering the DOM pools at equilibrium. The resulting quantity of carbon in vegetation was 31.2 t C/ha, comprised of 24.3 t C/ha in above-ground biomass (stem, bark, branch, foliage) and 6.9 t C/ha in below-ground biomass (coarse roots, fine roots). This compares to ranges of 6.5-130 t C/ha for above ground biomass reported for Canada West ((S&T)2 Consultants Inc. 2011)and 41.5 t C/ha average above ground biomass reported for Canada (Penman et al. 2003). The 31.2 t C/ha in vegetation on the land was assumed to result over a very long period of time, at equilibrium, in a quantity of carbon in soil (DOM pools) of 80 t C/ha. This can be confirmed by multiplying 31.2 t C/ha by the matrix in Table 3.6 (resulting in the matrix in Table 3.9), and then multiplying by the matrix in Table 3.7 (resulting in the matrix in Table 3.8). The resulting quantities of carbon for each of the DOM pools add to 80 t C/ha.

Table 3.9. Carbon content of live biomass pools from total live biomass carbon

	Live biomass carbon pools						
	M_{bm_stem}	M_{bm_bark}	M_{bm_branch}	$M_{bm\_foliage}$	$M_{bm\_coarse\_roots}$	$M_{bm\_fine\_roots}$	Total Biomass
Carbon mass [t C/ha]	14.9	2.7	5.2	1.4	5.9	1.1	31.2

3.3.5.4 Carbon soil loss at land-use change, above- and below-ground

The soil of lands prior to biomass production represent anywhere between 50-75% of the overall ecosystem carbon, and therefore can be an important carbon component (Gershenson *et al.* 2009). Converting existing lands for biomass production can rapidly release CO_2 from soil, as a result of burning and decomposition of leaves and fine roots, as well as slowly, via microbial decomposition of coarse roots and branches (Fargione *et al.* 2008). Land-use change activities typically include mechanical disturbances, which may accelerate decomposition by increasing the surface area of soil and cultivation (Paul *et al.* 2002).

Findings from other studies also suggest that soils with high carbon contents generally showed losses in carbon immediately following afforestation and in the first few years (5–10 years) afterwards, (Laganiere *et al.* 2010, Cowie *et al.* 2006, Paul *et al.* 2003, Paul *et al.* 2002, Samson *et al.* 1999, Hansen 1993, Kaul *et al.* 2010). Two recent studies investigating the soil carbon dynamics of hybrid poplar afforestation of marginal or former agricultural land (Mao *et al.* 2010, Zhang *et al.* 2010) report that organic C stocks in the soil decreased in the first 8-10 years following afforestation.

In terms of the magnitude of GHG emissions (CO_2 equivalent or C) resulting from direct land use conversion for biomass production for biofuel, Tyner *et al.* (2010) and

Searchinger *et al.* (2008) assumed a 25% loss of carbon in soil organic carbon in the top meter.

As such, we included the potential of initial carbon loss as an input parameter in the Bio-CarbD model, in order to be able to quantify the impact of such land-use changes on the carbon balance of the biofuel project.

For the test case we assumed a 12.5% C soil loss at land-use change. This assumption is viewed as conservative compared with other studies that considered up to twice as much carbon loss.

3.3.5.5 Loss of soil carbon from mechanized harvest activities

Harvest activities have the potential to significantly change soil carbon stocks (Gershenson *et al.* 2009). Depending on the type of harvest, tree species, rotation length, and the soils on which the forest is located, soil carbon stocks can experience anywhere from 40-60% declines to 20% gains. Some of this disturbance is an intentional part of site preparation, such as disking or plowing, and results in significant losses (over 20% of soil carbon) (Gershenson *et al.* 2009).

Nave *et al.* (2010) report 9% losses in mineral soil C following harvest, and 20% losses in surface mineral soil C when tillage is used following harvest. The authors also mention 36% losses in forest floor C at harvest-regeneration time (Fig. 2 and page 860). Cowie *et al.* (2006) report a reduction of 30 tC/ha in the soil, and 2 T C/ha for forest floor over a 100-year time frame for short rotation eucalypt.

Covington(1981) proposed a chronosequence model of forest floor carbon loss after harvest of up to 50% in the first 20 years. Some authors were not able to confirm the magnitude of the C loss (Yanai *et al.* 2003), however, still reported a similar trend of initial loss in the first 20 years after harvest.

Since harvest activities have the potential to significantly change soil carbon stocks, we included the loss of soil from mechanized harvest activities as an input to the Bio-CarbD model.

For the test case we assumed the values in Table 3.10 for loss of soil carbon due to the impact of mechanized harvest activities.

Table 3.10. Proportions of carbon losses from DOM pools from mechanized activities in harvest-planting years

	Proportion of carbon loss
DOM_ag_fast	7.50%
DOM_ag_slow	5.00%
DOM_ag_very_fast	10.00%
DOM_bg_fast	7.50%
DOM_bg_slow	5.00%
DOM_bg_very_fast	10.00%
DOM_medium	5.00%
DOM_sng_branch	5.00%
DOM_sng_stem	5.00%

3.3.6 Experimental scenarios for Bio-CarbD model

To demonstrate the Bio-CarbD model, the impacts of three variable elements were analyzed: (i) improvements in biomass yield and conversion efficiency, (ii) idle treatments allowed the end of the planning horizon; and (iii) initial carbon stocks in vegetation and soil, carbon losses from land-use change, and carbon losses from mechanized activities.

Since the Bio-CarbD carbon accounting model is linked to the C-BOS optimization model, the effects of improvements in both biomass yield and conversion efficiency can be analyzed. The impact on costs is analyzed with the C-BOS model, and the impact on carbon balance is analyzed with the Bio-CarbD model. To simplify the analysis and reduce the number of scenarios, only two situations were considered here in terms of improvements: either improvements were assumed for both biomass yield and conversion efficiency, or no improvements were assumed at all.

The impact of idle treatments on project carbon balance can also be analyzed with Bio-CarbD. In chapter 2 the C-BOS model runs showed that idle treatments resulted in lower project costs. By comparing scenarios with idle treatments vs. non-idle treatments, the effect on carbon sequestration can be investigated.

In order to investigate the effect of the initial carbon stocks and carbon losses due to the impact of mechanized activities on soil carbon, two situations were considered:

- the project was established on land with low carbon stocks, there was no carbon lost at land use change, and the mechanized activities had no effect on soil carbon loss,
- the land had high carbon stocks, land use change resulted in carbon loss, and mechanized activities also caused losses in soil organic matter carbon.

Each of these three elements was either present or absent in the scenarios, in order to enable the investigation of their potential effect on the resulting biogenic carbon balances. For example, the presence of idle treatments and improvements in biomass yield and conversion efficiency (scenarios 4 and 8) could result in a lower or higher project carbon balance compared with their absence, *i.e.* no improvements and no idle treatments (scenarios 1 and 5). The resulting scenario matrix is shown in Table 3.11.

Table 3.11. Scenario matrix

	No improvements in biomass yield nor in conversion efficiency		Improvements in both biomass and conversion efficiency	
	no idle treatments	idle treatments	no idle treatments	idle treatments
No initial C stocks in vegetation and soil; no losses from land-use change; no losses from mechanized harvest-planting activities	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Initial C stocks; losses from land-use change, losses from mechanized activities	Scenario 5	Scenario 6	Scenario 7	Scenario 8

For all scenarios we considered two land unit types, one non-irrigated (LU1) and one irrigated (LU2). For the non-irrigated land unit, LU1, two treatments were available, a single stem treatment S with rotation of 8 years and a coppice treatment S with 3 rotations of 5 years each (for a total 15 year rotation). For the irrigated land unit, LU2, two treatments were available, a single stem irrigated treatment Si with rotation of 6 years and a coppice irrigated treatment Si with 5 rotations of 3 years each (for a total 15 year rotation).

Similar to the assumptions of CHAPTER 2, the project planning horizon was set to 68 years. This compares to other policy-driven time scales, such as the time horizon of 100 years[3] now frequently chosen in policies and accounting related to the Kyoto Protocol (IPCC 2007, Fearnside 2002).

The assumed available planting area for each land unit was 100,000 ha. The biomass growth and yield parameters for each treatment are the same as those described earlier in CHAPTER 2. Non-production treatments at the beginning of the project were permitted, as well as idle treatments towards the end of the planning horizon if assumed in the respective scenarios.

The values for other model input parameters, including improvements in biomass yield and conversion efficiency, were the same as those in the test cases of CHAPTER 2, described in section 2.3.2.

For comparison with the findings from the previous chapter, the eight scenarios of CHAPTER 2 have been also analyzed from the net carbon balance perspective.

3.4 Results and discussion

3.4.1 Results from the C-BOS model

The results from the C-BOS model are shown in Table 3.12. As expected, the model produced the same solution to the optimization problem for the pairs of scenarios 1 and 5; 2 and 6; 3 and 7; and, 4 and 8; since the decision variables, the constraints and the formulation parameters were identical. The differences between the respective pairs of

[3] The project planning horizon, or the time period of assessment (the period over which GHG emissions and removals from a product system are considered), should not be confused with the characterisation time horizon (e.g. that for calculating global warming potentials – GWPs). According to Shine (2009), one of the lead authors who proposed the GWP concept in the IPCC First Assessment Report, the choice of the 100-year time horizon cannot be made on scientific grounds, but is a subjective, policy-driven, choice.

scenarios are related to biogenic carbon stocks and dynamics, which are not part of the optimization formulation.

Table 3.12. Results from the C-BOS model for scenarios 1-8

		Scenario 1&5	Scenario 2&6	Scenario 3&7	Scenario 4&8
Area LU1	[ha]	75,864	75,864	64,068	70,764
Area LU2	[ha]	0	0	966	15
Total Area	[ha]	75,864	75,864	65,034	70,779
Net avg. area LU1	[ha/yr]	71,959	68,054	61,167	53,880
Net avg. area LU2	[ha/yr]	0	0	899	7
Total Net Avg. Area	[ha/yr]	71,959	68,054	62,066	53,887
Objective function	[$ x10^6]	6,928	6,731	6,722	5,734
Harvested biomass (stem+bark +branch)	[t wood x10^6]	58	58	58	52
Total ethanol produced	[l x10^6]	13,855	13,855	15,409	13,855
Biomass production unit cost	[$/t wood]	119.38	115.98	116.46	110.96
Ethanol production unit cost	[$/l]	0.75	0.74	0.74	0.72

Consistent with the results in chapter 2, the idle scenarios resulted in lower project costs than the non-idle scenarios. The objective function of scenarios 2&6 was lower than that of scenarios 1&5; similarly, the objective function of scenarios 4&8 was lower than that of scenarios 3&7. The project costs are reduced by not planting unnecessary hectares and not undertaking activities on land parcels that will not be harvested by the end of the project.

3.4.2 Project life cycle biogenic carbon balance

The Bio-CarbD model calculates the carbon balance for each year of the project, by individual carbon pools. Table 3.13 through Table 3.20 show the carbon balance of the respective carbon pools: 1) before land-use change (column “Prior to LUC”); 2) immediately after land-use change (column “Year -1 (LUC)”); 3) at the end of the project (column “Year 68”); and, 4) the life cycle project balance (last column).

In scenarios 1 through 4, the absolute value of C_ABSORBED is in fact the sum of the carbon pools at the end of the project (Year 68). The carbon balances for all pools in year 68 are the same as the project life cycle balances, since there are no initial quantities of organic material in vegetation and DOM before land-use change.

The carbon balance values in scenario 1 (Table 3.13) can be interpreted as follows: 58,336,113 t C were absorbed from the atmosphere during the project planning horizon. At the end of the project, 2,334,972 t C were sequestered in live biomass pools (*i.e.* existing on site at end of year 68 after the project ends), 5,908,078 t C were sequestered in DOM pools, 29,014,756 t C were harvested, and 21,078,308 t C were released to the atmosphere as a result of decomposition of dead organic matter.

For the purposes of the carbon balance analysis in this study, the quantity of harvested biomass was assumed to become non-biogenic (with part of it being converted to ethanol in the same year, and part of it being just losses through various processing steps) and therefore considered as an emission from the perspective of the biofuel project. While it is true that some of the harvested biomass will be converted to ethanol, which displaces gasoline, that part of the carbon cycle is not the subject of the carbon balance calculations here. The emissions and credits from the conversion of wood to ethanol and from the final combustion of ethanol in internal combustion engines will be considered explicitly in CHAPTER 4, where all the project non-biogenic carbon sources, as well as the appropriate carbon credits, will be accounted for. In this chapter, it is considered that the carbon contained in the harvested biomass is not sequestered in live biomass, it is also not transferred to DOM pools, and therefore can be considered as equivalent to an emission.

The detailed dynamics of the carbon accumulation in live biomass and dead organic matter pools for all scenarios are shown in APPENDIX E. For an even more detailed view of the carbon dynamics at the individual parcel level, a few examples of non-irrigated and irrigated parcels are given in APPENDIX F.

Table 3.13. Carbon balance in selected carbon pools, scenario 1 [t C]

SCENARIO 1		Prior to LUC	Year -1 (LUC)	Year 68	Life cycle project balance
C_SEQUESTERED	C_SEQ_LBIO	0	0	2,334,972	8,243,050
	C_SEQ_DOM	0	0	5,908,078	
C_RELEASED	C_HBIO	0	0	29,014,756	50,093,063
	C_DCAY	0	0	21,078,308	
	C_ ABSORBED	0	0	-58,336,113	-58,336,113
TOTAL		0	0	0	0

Table 3.14. Carbon balance in selected carbon pools, scenario 2 [t C]

SCENARIO 2		Prior to LUC	Year -1 (LUC)	Year 68	Life cycle project balance
C_SEQUESTERED	C_SEQ_LBIO	0	0	0	6,637,780
	C_SEQ_DOM	0	0	6,637,780	
C_RELEASED	C_HBIO	0	0	29,014,756	48,646,779
	C_DCAY	0	0	19,632,024	
	C_ ABSORBED	0	0	-55,284,559	-55,284,559
TOTAL		0	0	0	0

Table 3.15. Carbon balance in selected carbon pools, scenario 3 [t C]

SCENARIO 3		Prior to LUC	Year -1 (LUC)	Year 68	Life cycle project balance
C_SEQUESTERED	C_SEQ_LBIO	0	0	2,762,557	7,858,285
	C_SEQ_DOM	0	0	5,095,728	
C_RELEASED	C_HBIO	0	0	28,858,085	45,945,231
	C_DCAY	0	0	17,087,146	
	C_ ABSORBED	0	0	-53,803,517	-53,803,517
TOTAL		0	0	0	0

Table 3.16. Carbon balance in selected carbon pools, scenario 4 [t C]

SCENARIO 4		Prior to LUC	Year -1 (LUC)	Year 68	Life cycle project balance
C_SEQUESTERED	C_SEQ_LBIO	0	0	0	6,236,902
	C_SEQ_DOM	0	0	6,236,902	
C_RELEASED	C_HBIO	0	0	25,838,982	41,706,743
	C_DCAY	0	0	15,867,761	
	C_ ABSORBED	0	0	-47,943,645	-47,943,645
TOTAL		0	0	0	0

In scenarios 5-8 the calculations are somewhat more complicated, since we need to account for the initial carbon quantities in vegetation and DOM before land-use change, and include them in the carbon dynamics over time, as well as in the overall project life cycle.

In scenario 5, the carbon in live biomass pools (both above- and below-ground) that existed on the land units prior to land-use change totalled 2,365,859 t C (the C_SEQ_LBIO pool). This was calculated as the sum of: the initial above-ground vegetation (24.3 t C/ha * 75,863.61 ha = 1,839,692.54 t C); the initial coarse roots biomass (18.78% * 31.19 t C/ha * 75,863.61 ha = 444,342.43 t C); and the initial fine rots biomass (3.46% * 31.19 t C/ha * 75,863.61 ha = 81,823.98).

The initial carbon in dead organic matter pools prior to land-use change in scenario 5 (the C_SEQ_DOM pool) was 6,058,890 t C (calculated from 80 t C/ha * 75,863.61 ha).

Scenarios 5 through 8 assumed carbon losses during land-use change. These losses were calculated from the initial carbon quantities and the rate of loss.

For example, in scenario 5 there was zero carbon left in live biomass pools after land-use change (the C_SEQ_LBIO pool in column labelled "Year -1 LUC" in Table 3.17). The live biomass carbon was transferred/removed as follows:

- the foliage (107,295 t C = 2,365,859 t C * 4.54%) was assumed to be transferred 100% to the dom_ag_very_fast pool as per the carbon transfer matrix applied to all treatments only once at land-use change (Table 3.5)
- the stem, bark, and branch biomass (1,732,398 t C = 2,365,859 t C * (47.76% + 8.67% + 16.79%)) was assumed to be removed from the site as part of the land-use change operations, and therefore not included in these calculations
- the coarse roots biomass was transferred 50% to the dom_ag_fast and 50% to the dom_bg_fast pools, as per the carbon transfer matrix in Table 3.5
- the fine roots biomass was transferred 50% to the dom_ag_very_fast and 50% to the dom_bg_very_fast pools, as per the carbon transfer matrix in Table 3.5

The carbon quantity in DOM pools after land-use change (the C_SEQ_DOM pool in column "Year -1 LUC" in Table 3.17) was calculated from the initial carbon in DOM

(6,058,890 t C), the proportions of carbon in DOM pools in Table 3.8, and the carbon transfer matrix coefficients from DOM pools to DOM pools in Table 3.5. As a result of these transfers/removals, the carbon in the C_SEQ_DOM pool after land-use change was calculated to be 5,773,651 t C.

The carbon released through decay after land-use change (the C_DCAY pool in column "Year -1 LUC" in Table 3.17) was calculated from the initial carbon in DOM (6,058,890 t C), the proportions of carbon in DOM pools in Table 3.8, and the carbon transfer matrix coefficients from DOM pools to CO2_dom in Table 3.5. The carbon in the C_DCAY pool after land-use change was calculated to be 918,700 t C.

The life cycle carbon balance in the C_SEQUESTERED carbon pool is the difference between a) the sum of C_SEQ_LBIO and C_SEQ_DOM pools in Year 68, and b) the sum of C_SEQ_LBIO and C_SEQ_DOM pools prior to LUC. For example, in scenario 5: (2,334,972 + 7,469,791) – (2,365,859 + 6,058,890) = 1,380,014. The positive sign of the carbon quantity in the C_SEQUESTERED pool signifies that at the end of the project there was more carbon sequestered in live biomass and DOM pools than before land-use change. In other words, the presence of the biofuel project in scenario 5 resulted in an increase of the net sequestered carbon compared with the absence of the project.

In contrast, in scenario 6 the presence of the project resulted in a reduction of the net sequestered carbon compared with the absence of the project (assuming that in the absence of the project the quantity of sequestered carbon was the same throughout the planning horizon, as mentioned in section 3.3.5.2). Since the idle treatments were allowed in scenario 6 compared with scenario 5, land parcels were "abandoned" towards the end of the project unless they could be harvested, and therefore no carbon was sequestered in the live biomass pools at the end of the project (C_SEQ_LBIO was 0 in year 68, Table 3.18). Also due to the idle treatments, less plantation area was used and less carbon was absorbed from the atmosphere (3,051,554 t C less than in scenario 5; pool C_ABSORBED). As a result, less carbon was released from DOM decay (1,395,383 t C less than in scenario 5; pool C-DCAY), and less carbon was sequestered in live biomass (2,334,972 t C less than scenario 5; pool C_SEQ_LBIO). The quantity of carbon sequestered at the end of the project in scenario 6 was 8,148,592 t C, which was less than the initial carbon before the project

8,424,749 t C; the difference was 276,157 t C, which is the negative balance shown in the C_SEQ_LBIO pool in Table 3.18. In scenario 6 the biofuel project resulted in a negative carbon balance, which is not a desirable outcome from a climate change mitigation perspective.

The negative project carbon balance in scenario 6, compared with the positive carbon balance in scenario 5, suggests that the allowance of idle treatments resulted in less carbon sequestered at the end of the project.

It is important to note that these results must be interpreted in the context of the fact that in this chapter we only account for the biogenic carbon of the biomass production system, and we do not consider the substitution credits for displacement of gasoline (also known as fuel switch credits). These elements will be considered next in CHAPTER 4, where a full life cycle carbon and GHG analysis will be considered.

The model Bio-CarbD monitors the biogenic carbon in pools starting with the moment just after land-use change, when the carbon in the C-DOM and C_DCAY pools represent the changes caused by the land-use change. Then, throughout the years, it accounts for the carbon absorbed from the atmosphere, the carbon sequestered in biomass and DOM, and the carbon emitted back to the atmosphere. In Table 3.13 through Table 3.20 the total quantity of carbon in each of the first three columns (columns Prior to LUC, Year -1 LUC and Year 68) is constant within each scenario; this is a direct representation of the carbon mass conservation used in the Bio-CarbD model calculations.

The life cycle carbon balance of the C_RELEASED carbon pools is the sum of the C_HBIO and C_DCAY pools in year 68, since the end-of-project carbon dynamics calculations already include the land-use change quantities.

The results for scenarios 6-8 are shown in Table 3.18 through Table 3.20 and are determined similarly with the calculations for scenario 5.

Table 3.17. Carbon balance in selected carbon pools, scenario 5 [t C]

SCENARIO 5		Prior to LUC	Year -1 (LUC)	Year 68	Life cycle project balance
C_SEQUESTERED	C_SEQ_LBIO	2,365,859	0	2,334,972	1,380,014
	C_SEQ_DOM	6,058,890	5,773,651	7,469,791	
C_RELEASED	C_HBIO	0	1,732,398	30,747,154	56,956,099
	C_DCAY	0	918,700	26,208,945	
	C_ ABSORBED	0	0	-58,336,113	-58,336,113
TOTAL		8,424,749	8,424,749	8,424,749	0

Table 3.18. Carbon balance in selected carbon pools, scenario 6 [t C]

SCENARIO 6		Prior to LUC	Year -1 (LUC)	Year 68	Life cycle project balance
C_SEQUESTERED	C_SEQ_LBIO	2,365,859	0	0	-276,157
	C_SEQ_DOM	6,058,890	5,773,651	8,148,592	
C_RELEASED	C_HBIO	0	1,732,398	30,747,154	55,560,716
	C_DCAY	0	918,700	24,813,563	
	C_ ABSORBED	0	0	-55,284,559	-55,284,559
TOTAL		8,424,749	8,424,749	8,424,749	0

Table 3.19. Carbon balance in selected carbon pools, scenario 7 [t C]

SCENARIO 7		Prior to LUC	Year -1 (LUC)	Year 68	Life cycle project balance
C_SEQUESTERED	C_SEQ_LBIO	2,028,135	0	2,762,557	2,360,573
	C_SEQ_DOM	5,193,990	4,949,468	6,820,141	
C_RELEASED	C_HBIO	0	1,485,100	30,343,185	51,442,943
	C_DCAY	0	787,557	21,099,758	
	C_ ABSORBED	0	0	-53,803,517	-53,803,517
TOTAL		7,222,124	7,222,124	7,222,124	0

Table 3.20. Carbon balance in selected carbon pools, scenario 8 [t C]

SCENARIO 8		Prior to LUC	Year -1 (LUC)	Year 68	Life cycle project balance
C_SEQUESTERED	C_SEQ_LBIO	2,207,302	0	0	203,123
	C_SEQ_DOM	5,652,832	5,386,708	8,063,257	
C_RELEASED	C_HBIO	0	1,616,295	27,455,277	47,740,522
	C_DCAY	0	857,130	20,285,245	
	C_ ABSORBED	0	0	-47,943,645	-47,943,645
TOTAL		7,860,134	7,860,134	7,860,134	0

To demonstrate the methodology proposed in this chapter, the project life cycle biogenic carbon balances shown in Table 3.21 were compared between scenarios in three ways, referring to the scenario matrix of Table 3.11:

1. compare the yield improvement scenarios with the non-improvement scenarios: *i.e.* scenario 3 with 1, scenario 4 with 2, scenario 7 with 5, and scenario 8 with 6;
2. compare the idle treatment scenarios with the non-idle scenarios: *i.e.* scenario 2 with 1, scenario 4 with 3, scenario 6 with 5, and scenario 8 with 7;
3. compare the zero initial carbon stocks scenarios with the non-zero scenarios: *i.e.* scenario 5 with 1, scenario 6 with 2, scenario 7 with 3, and scenario 8 with 4.

Table 3.21. Net biogenic carbon balance by scenario [t C] (the quantities shown in brackets represent the carbon balance per hectare, [t C/ha])

	No improvements in biomass yield nor in conversion efficiency		Improvements in both biomass and conversion efficiency	
	no idle treatments	idle treatments	no idle treatments	idle treatments
No initial C stocks in vegetation and soil; no losses from land-use change; no losses from mechanized harvest-planting activities	Scenario 1 8,243,050 (115)	Scenario 2 6,637,780 (98)	Scenario 3 7,858,285 (127)	Scenario 4 6,236,902 (116)
Initial C stocks; losses from land-use change, losses from mechanized activities	Scenario 5 1,380,014 (19)	Scenario 6 -276,157 (-4)	Scenario 7 2,360,573 (38)	Scenario 8 203,123 (4)

3.4.2.1 Comparison of yield improvement and non-improvement scenarios

The first comparison is between scenarios 3 and 1. As seen in Table 3.21, scenario 3 sequestered less tonnes of net biogenic carbon than scenario 1 (7,858,285 t C < 8,243,050 t

C). This may seem counterintuitive at first, because if the yields were improved in the future, then it would be expected that the project would result in a higher quantity of carbon sequestered. However, it is important to note that scenario 3 absorbed less carbon from the atmosphere because it needed to produce less biomass (due to the improved conversion efficiency), and also released less carbon due to DOM decay (due to using less hectares and having less carbon being absorbed from atmosphere and subsequently transferred to the DOM pools). This in fact means that, if the two scenarios are compared on a relative basis, the opposite conclusion is reached: scenario 3 sequestered 14.6% of the carbon absorbed from the atmosphere (7,858,285 ÷ 53,803,517), more than scenario 2 which sequestered only 14.1%. In other words, even though scenario 3 sequestered a larger proportion of the carbon absorbed from the atmosphere than scenario 1, it sequestered a smaller net quantity of carbon.

This suggests that the quantity of carbon absorbed from the atmosphere is an important component of the project carbon balance, and its omission could result in incorrect conclusions. For example, if the biomass converted to ethanol would be simply assumed "carbon-neutral" then the carbon absorbed from the atmosphere would not be accounted for, the carbon sequestered in DOM pools and live biomass would also not be accounted for, so the benefits of the biofuels project in terms of the biogenic carbon (*i.e.* the positive net carbon sequestered at the end of the project) would not be detected.

The plantation area may also contribute to the carbon calculations, when the impacts are evaluated on a per hectare basis. Since the annual quantity of ethanol needed is assumed to be constant in this study (*i.e.* one biorefinery producing the same volume of ethanol each year), the scenarios that assumed improvements in biomass yield and conversion efficiency required less plantation hectares, as seen in Table 3.12. For example, scenario 3 required a total net average area of only 61,167 ha, compared with the 71,959 ha required in scenario 1. On a per hectare basis, scenario 3 sequestered more carbon than scenario 1, *i.e.* 127 t C/ha *vs*. 115 t C/ha.

From a carbon balance perspective, scenario 3 does not seem to be a desirable strategy compared with scenario 1. Even though scenario 3 has sequestered more carbon than scenario 1 on a per hectare basis and as a proportion of the carbon absorbed from the atmosphere, it still sequestered a lower overall quantity of carbon. In scenario 3 less hectares were used because of the improvements in biomass yield and conversion efficiency, and this in turn has negated the benefits of sequestering carbon in live biomass and DOM pools on the land (which have been greater in scenario 1 where more hectares were used). In both scenarios the project carbon balance was improved because carbon was sequestered on the land (in biomass and DOM) at the end of the project; the larger the plantation area, the more carbon was sequestered. This explains why scenario 3, which used less hectares, sequestered a smaller net quantity of carbon than scenario 1, causing a negative impact on the project carbon balance.

The second comparison, of scenarios 4 and 2, results in a similar conclusion: scenario 4 sequestered less net carbon (6,236,902 t C) than scenario 2 (6,637,780 t C) from a net quantity perspective. However, scenario 4 sequestered 13.0% of the carbon absorbed from atmosphere (6,236,902 ÷ 47,943,645) compared with scenario 2 which sequestered only 12.0%; scenario 4 also sequestered more carbon on a per hectare basis than scenario 2: 116 t C/ha *vs*. 98 t C/ha.

In short, the improvements in biomass yield and conversion efficiency had a negative impact on the net carbon sequestered by the low-carbon scenarios 1 through 4.

In contrast, in the other two comparisons (scenario 7 *vs*. 5, and 8 *vs*. 6, where the initial C stocks were greater than zero, and where carbon losses occurred from land-use change and from the effect of mechanized activities on soil), the yield improvement scenarios have sequestered more carbon than the no-improvement scenarios. It is important to note that the project carbon balance of scenarios 5-8 was much less than of scenarios 1-4. The combination of the initial carbon stocks and the losses have resulted in much less carbon being sequestered at the end of the project; in the case of scenario 6 not only there was no net carbon sequestered at end of project, but the project actually lost more carbon that it absorbed from the atmosphere. Looking at the effect of improvements, scenario 7 sequestered more net carbon than scenario 5 by all calculations, in terms of the carbon quantity, the proportion of carbon sequestered from the carbon absorbed, and on a per hectare basis. The same result was observed for scenarios 8 and 6. Again, the size of plantation area can explain why the net carbon sequestration was higher in the yield improvement scenarios: since carbon losses from a high initial carbon stock were assumed, the scenarios that had a smaller plantation area incurred less carbon losses. For example, scenario 7 had a total net average area of 68,054 ha/yr, less than the 71,959 ha/yr of scenario 5. Prior to land-use change, the plantation area of scenario 7 had 7,222,124 t C in live biomass and DOM, less than the 8,424,749 t C of scenario 5. Proportionally, both scenarios lost carbon at land-use change at the same rate, 31.5% (comprised of 100% loss of carbon in live biomass and 4.7% loss of carbon in DOM pools). However, the net tonnes of carbon lost at land-use change in scenario seven, 2,272,657 t C, were less than the 2,651,098 t C lost in scenario 5. This, combined with the facts that a) scenario 7 absorbed much less carbon from the atmosphere than scenario 5 (because it did not need to produce as much biomass due to the improvements in conversion efficiency), b) it emitted much less carbon from DOM decay (21,099,758 t C *vs*. 26,208,945 t C), and c) it accumulated almost as much carbon in live and DOM pools (9,582,698 t C *vs*. 9,804,763 t C), resulted in a higher net carbon sequestered by scenario 7 than scenario 5 (2,360,573 t C *vs*. 1,380,014 t C).

In short, the improvements in biomass yield and conversion efficiency had a positive impact on the net carbon sequestered by the high-carbon scenarios 5 through 8. This is in contrast with the results from the low-carbon scenarios 1 through 4, where the improvements had a negative impact on the net carbon sequestered.

A conclusion suggested by these results is that the future improvements in biomass yield and conversion yield can result in either negative impacts on the project net carbon balance (for low-carbon projects) or in positive impacts (for high-carbon projects).

From a project cost perspective, all the yield improvement scenarios have resulted in lower project costs, as expected. This is a desirable outcome for biofuels production projects, which pursue cost minimization strategies.

The trade-offs between project costs and sequestered carbon can be calculated for each pair of scenarios. For example, in scenario 3 the production cost ($6,721,517,384) was lower than in scenario 1 ($6,927,745,887), but the carbon sequestered (7,858,285 t C) was also lower than in scenario 1 (8,243,050 t C). The difference between the cost/carbon ratios for the two scenarios was $536/t C ($206,228,503 ÷ 384,764 t C).

This means that, if the biofuels project would be able to sell carbon credits on account of the net carbon sequestered, the price of carbon would need to be at least $536/t C in order to entice the project proponent to pursue scenario 1 instead of scenario 3. To summarize for the low carbon and low carbon impact scenarios:

- A carbon price of at least $536/t C is needed to pursue scenario 1 instead of 3,
- $2,485/t C for scenario 2 instead of 4.

Since these carbon prices are much higher than any current market prices for carbon offsets, it seems imperative to continue the research efforts that can lead to improvements in biomass yield and conversion efficiency.

In conclusion, the improvement in biomass yield and conversion efficiency does not necessarily result in more carbon sequestration: on one hand, in the low carbon and low carbon impact scenarios (1 through 4) the yield improvements actually resulted in less carbon being sequestered; one the other hand, in the high carbon and high carbon impact scenarios (5 through 8) the yield improvements have resulted in more biogenic carbon sequestered by the biofuel project.

3.4.2.2 Comparison of idle and non-idle treatment scenarios

As expected, allowing the idle treatments has reduced the ethanol production cost from $0.75/l to $0.74/l in the no yield improvement scenarios, and from $0.74/l to $0.72/l in the yield improvement scenarios (Table 3.12). However, this has also decreased the carbon sequestered by the idle treatments scenarios. The carbon sequestered in the idle treatment scenario 2 was lower than that of the no idle treatment scenario 1 (6,637,780 t C < 8,243,050 t C). Similarly, the carbon sequestered in scenario 4 was lower than in scenario 3, in scenario 6 lower than 5, and in scenario 8 lower than 7 (Table 3.21).

The Bio-CarbD model results suggests that, even though the idle treatments reduced project costs, they also resulted in less net carbon sequestered over the life cycle of the project in all the cases analyzed. The trade-offs for all scenarios can be summarized as follows:

- A carbon price of at least $123/t C ($197,222,189 ÷ 1,605,270 t C) is needed in order to entice the project proponent to pursue scenario 1 instead of 2,

- \$609/t C for scenario 3 instead of 4,
- \$119/t C for scenario 5 instead of 6,
- \$458/t C for scenario 7 instead of 8.

3.4.2.3 Comparison of zero and non-zero initial carbon stocks, losses and non-losses of carbon, from land-use change and from mechanized activities scenarios

The carbon sequestered in the “high carbon” scenarios 5 through 8 (which exhibited initial carbon stocks, losses from land-use change, and losses from mechanized activities) was considerably lower than that of the corresponding “low carbon” scenarios 1 through 4 (with no initial carbon stocks or losses). The carbon sequestered in scenario 5 was lower than scenario 1 (-159,586 t C < 8,243,050 t C); scenario 6 was lower than 2; scenario 7 was lower than 3; and scenario 8 was lower than 4 (see Table 3.21).

In scenario 6 the quantity of sequestered carbon was actually negative. The accumulation of carbon in the DOM pools by the end of the project was not sufficient to make up for the initial carbon losses in live biomass and DOM pools. This means that the presence of the biofuel project has produced more carbon emissions and less carbon sequestration than the absence of the project.

This suggests that initial carbon stocks and carbon losses from project activities had a large effect on the quantity of carbon that was sequestered by the biofuel project. This finding warrants further exploration of these effects separately for initial carbon stocks, for losses from land-use change, and for losses from mechanized activities, to determine the importance of each of the effects. This analysis will be examined in CHAPTER 4 and CHAPTER 5.

It is important to note that for each pair of scenarios that are compared in this section (as seen in Table 3.12: scenario 1 with 5; 2 with 6; 3 with 7; and, 4 with 8) there is no difference in the C-BOS optimization model results between the two scenarios. This means that, regardless of the quantities of carbon in initial C stocks and the losses from land-use change and from impact on soil carbon of mechanized activities, the project production costs will be the same for the two projects; this is a direct consequence of the mathematical formulation of the C-BOS optimization model which has the objective to minimize project costs, and is non-cognizant of biogenic carbon dynamics (which are calculated *a posteriori* by the Bio-CarbD model).

3.4.3 Analysis of scenarios from chapter 2

For comparison with the findings from the previous chapter, the eight scenarios of CHAPTER 2 have been also analyzed from the net carbon balance perspective.

The largest amount of net biogenic carbon sequestered at the end of the project was in Scenario 2B, where only improvements in biomass yield were considered, and where no idle treatments were allowed (Table 3.22). This is in contrast with the findings in Chapter 2, where the preferred scenario was 4A.

The lowest ethanol production cost ($ 0.81/l in scenario 4A) resulted in a net 6,728,817 tonnes of carbon sequestered. A higher production cost ($0.90/l in Scenario 2B) would result in more carbon sequestered, i.e. 2,010,984 tonnes more. In other words, from scenario 4A to scenario 2B the production cost would increase by $1,508,138,589 and the carbon sequestered would increase by 2,010,984 tonnes of carbon. This is equivalent to a differential cost of $750/tonne of carbon, which makes scenario 2B unlikely to be pursued as a carbon sequestration strategy.

It is important to note that the biogenic carbon balance do not include the GHG emissions generated by the bioenergy project, not the credits from displacing gasoline. This will be the subject of CHAPTER 4.

Table 3.22. C-BOS and BioCarbD results for the eight scenarios of CHAPTER 2

		Scenario 1A	Scenario 2A	Scenario 3A	Scenario 4A	Scenario 1B	Scenario 2B	Scenario 3B	Scenario 4B
C-BOS objective function value	[$ x10^6]	8,222	7,869	7,327	7,033	8,441	8,460	7,905	8,220
Ethanol unit production cost	[$/l]	0.90	0.87	0.83	0.81	0.91	0.90	0.86	0.84
Total ethanol produced	[l x10^6]	13,855	13,855	13,855	13,855	13,855	14,122	14,083	15,411
Average annual area	[ha/yr]	79,873	71,079	70,997	63,947	84,456	75,606	75,593	72,635
Net biogenic C sequestered	[t C x10^6]	7.05	7.33	6.48	6.73	8.70	8.74	7.33	8.27

3.5 Limitations of the study

One of the limitations of the study is the reference of CBM-CFS3 parameters (*i.e.* carbon transfer matrix coefficients, transfer and decay parameters) specified for hardwood species, which are assumed to be suitable for this study. This aspect could be improved in the future as new research with more appropriate parameters is published.

The scenarios described in this study were generated using only three input variables: yield improvement, idle treatments, initial carbon stocks and losses. This could be considered a limitation for the carbon stock-related changes, for example, because all these effects were amalgamated. This can be addressed, however, by considering separately the impacts of 1) initial carbon stocks, 2) carbon losses at land-use change, and 3) carbon losses from mechanized activities, on the project life cycle carbon balance. Similarly for the yield improvements, the separate effect of biomass yield and of conversion efficiency could be investigated. This type of analysis will be considered in CHAPTER 5.

An important component of planning biomass production over a long time horizon is uncertainty. Life-cycle greenhouse-gas-performance estimates of second-generation biofuels remain uncertain in the absence of large-scale crop production trials and commercial-scale biorefineries (IEA 2010). The Bio-CarbD model described in this chapter assumes that the biomass growth and yield, and the conversion efficiency are known. An important area for further research is to investigate the effect of uncertainty on biomass production planning, which is explored using sensitivity analysis in CHAPTER 5.

3.6 Conclusion

This chapter describes the life cycle biogenic carbon balance and dynamics model (Bio-CarbD), which determines the life cycle biogenic carbon balance of a biofuel project and the dynamics of biogenic carbon pools throughout the planning horizon on an annual basis. The model is based on a mass balance methodology for quantitative analysis of the life cycle biogenic carbon transfer and dynamics between the atmosphere, live biomass pools, and dead organic matter pools, at the project level. Bio-Carbd includes and monitors individually all the carbon pools that are included in the CBM-CFS3 model, compared with other approaches which did not include the slow and very slow DOM pools, and combined the remaining pools into four amalgamated pools (Hennigar *et al.* 2008, Neilson *et al.* 2008).

The main outcomes of the Bio-CarbD model are: 1) the life cycle biogenic carbon balance of the biofuel project; 2) the dynamics of biogenic carbon pools throughout the planning horizon on an annual basis.

The Bio-CarbD model is a key component of the Carbon Balance and Biomass to Biofuel Optimization planning system (C3BO) described in CHAPTER 4, and is further used in the research work of CHAPTER 5 and CHAPTER 6.

One innovative approach of the Bio-CarbD model is that it does not make any assumptions about the “carbon-neutrality” of the biomass generated by the biofuel project. Instead, Bio-CarbD accounts for all the biogenic carbon absorbed from the atmosphere through natural biomass growth processes, carbon transfers between live biomass pools and dead organic matter pools, carbon emissions from biodegradation of organic material through both DOM decay and biomass to biofuel conversion processes, as well as carbon emissions from the end use of the ethanol at the end of the life cycle.

A novelty of the implementation of the “one inventory plus change” methodology in Bio-CarbD is that the carbon pools are defined and monitored within individual land parcels, which are kept intact throughout the planning horizon (*i.e.* hectares of one land planning unit are not broken up and combined with hectares from other planning units that are harvested at the same time). Other adaptations of the CBM-CFS3 method (Hennigar *et al.* 2008, Neilson *et al.* 2008) use a forest management planning method that is designed to allow for breaking up the hectares of land units and combining them with hectares from other land units that are being harvested in the same year. However, those hectares that are broken up and combined may have very different carbon pools, even if they are harvested at the same time. When these different size carbon pools are combined together the information about the actual carbon quantity in the resulting hectares is lost.

The scenario comparisons presented in this chapter indicates that all three categories of scenario inputs (improvements in biomass yield and conversion efficiency, idle treatments, and initial carbon stocks and soil carbon losses from mechanized activities) affect the project biogenic carbon balance.

First, the improvements in biomass yield and conversion efficiency had contradicting effects: it both decreased the net carbon sequestered (in the zero initial carbon and carbon impact scenarios) and increased the net carbon sequestered (in the non-zero initial carbon and carbon impact scenarios). This was explained mainly by the size of the plantation area. Since the yield improvements led to a decrease in plantation area, the net quantity of carbon sequestered in the yield improvement scenarios was: a) smaller when the land was accumulating carbon in live biomass and DOM pools (in the zero initial carbon and carbon impact scenarios), and b) larger when the land was "losing" carbon through land-use change and other impacts (in the non-zero initial carbon and carbon impact scenarios).

From a project cost perspective, the improvements in biomass yield and conversion efficiency resulted in lower costs, as expected. In the cases when the yield improvements also resulted in lower sequestered carbon, the opportunity cost of carbon for the yield improvement scenarios was between \$536/t C and \$2,485/t C compared to the no improvement scenarios.

Second, allowing idle treatments (towards the end of the planning horizon) resulted in all cases in less carbon sequestered over the life cycle of the project. The opportunity cost for carbon in the idle treatment scenarios was between \$123/t C and \$609/t C compared with the no idle treatment scenarios.

Finally, the scenario comparisons suggest that the initial carbon stocks and carbon losses or emissions due to land-use change are important to the carbon balance of biofuel projects. This outcome held in our analysis even in the case when the biomass yield and conversion efficiency improved over time. Therefore, this suggests that life-cycle analyses can be greatly improved if these factors are considered. This prompts a need of further research into the sensitivity analysis of the effects of the three variables mentioned above, which will be the subject of CHAPTER 5.

CHAPTER 4 C3BO: a method for assessing the greenhouse gas and carbon balance of biofuel projects that displace gasoline with wood ethanol from fast growing tree plantations

4.1 Synopsis

From the perspective of greenhouse gas and carbon balances, under what conditions can projects that displace gasoline with wood ethanol from fast growing tree plantations be viable climate mitigation strategies? This chapter introduces the Carbon Balance and Biomass to Biofuel Optimization planning model (C3BO), a model that proposes to answer this question by comparing: the overall life-cycle greenhouse gas and carbon balance of a biofuel project that establishes tree plantations, builds a biorefinery that produces ethanol from the wood biomass, uses all the ethanol in internal combustion engines; with the GHG balance of the displaced gasoline production system.

The C3BO model combines the optimal biomass production strategies determined by the Carbon-aware Biomass Production Optimization System (C-BOS) developed in CHAPTER 2, with the carbon balance calculations in the Biogenic Carbon Dynamics Model (Bio-CarbD) developed in CHAPTER 3.

In addition to the biogenic carbon pools stocks and fluxes determined in the Bio-CarbD model, C3BO also considers the overall project greenhouse gas balances:

- GHG emissions from use of fossil fuels (diesel) by biomass production machinery and biomass transportation trucks,
- GHG emissions from biorefinery activities to convert project biomass into ethanol,
- Carbon and GHG emissions from initial direct land-use change,
- Credits for net carbon sequestered on site at the end of planning horizon, in live biomass and in DOM pools,
- Credits for the recovered lignin that is used in energy production displacing fossil fuel,
- Credits for avoided emissions of the gasoline being displaced by ethanol.

To illustrate the use of the C3BO model for analyzing biofuel project-specific GHG balances, four scenarios are proposed with low and high ethanol production costs, and low and high GHG ethanol emissions. Each scenario uses a prototype biomass production system and biorefinery, and the C3BO model calculates the life cycle GHG balance of the ethanol production system and compares this with that of the displaced gasoline (the baseline or business-as-usual case).

C3BO model results indicate that, under certain conditions, ethanol production system emissions are higher than those of the displaced gasoline production system, and under other conditions they are not. In both scenarios with assumed high GHG ethanol emissions, net ethanol GHG emissions were substantially higher than those of the displaced gasoline, which implies that the fuel switch would not be a viable mitigation strategy under these conditions. However, in both scenarios with assumed low GHG ethanol emissions, the ethanol project emissions were lower than those of gasoline.

The key conclusion of this study is that displacing gasoline with wood ethanol from fast growing tree plantations may not always be a viable climate mitigation strategy; however, many conditions seem to exist where the fuel switch can indeed be viable. We suggest that the question of the viability of the fuel switch cannot be answered with a generic predisposition that “wood ethanol is lower in GHG emissions than gasoline” or “always higher GHG emissions than gasoline.” In short, it depends on project-specific input parameters, and therefore the viability must be determined on a project-by-project basis.

4.2 Introduction

Emissions of greenhouse gases (GHG) from the production and consumption of fossil fuels represent over 56% of anthropogenic GHG emissions, which are influencing the recent global warming trend (IPCC 2007). An essential strategy for mitigating climate change would be to reduce the consumption of fossil fuels, but it is unrealistic to expect liquid transportation fuel consumption to decrease over the next decades (IEA 2010). It seems therefore important to develop sustainable alternatives to fossil-derived transportation fuels.

One way to accomplish this is to substitute fossil-derived transportation fuels with biofuels produced from renewables such as wood biomass. Dedicated energy plantations of fast growing poplar trees have been identified as a key potential source of woody biomass for production of biofuel. Biofuel produced from short-rotation tree plantations has the potential to both reduce the overall atmospheric GHG emissions and increase terrestrial carbon (C) stocks, compared with the displaced fossil fuel, since biomass absorbs CO_2 from the atmosphere and sequesters it when is grown on a sustained cycle.

However, the effectiveness of large-scale wood-to-energy projects is still not well understood in terms of their life-cycle greenhouse gas balances and carbon benefits and their impact on the land base. There is broad agreement in the scientific community that Life-Cycle Assessment (LCA) is one of the best methodologies for the GHG balance calculation of biomass based energy systems (Gnansounou *et al.* 2009, Cherubini 2010). However, wide ranges of greenhouse gas balances calculations are reported in the peer-reviewed literature on GHG balance analyses for lignocellulosic biofuels projects, where different methods are employed for carbon accounting. This, therefore, results in conflicting results on these projects' impacts on atmospheric and terrestrial carbon stocks (Larson 2006, von Blottnitz and Curran 2007, Searchinger *et al.* 2008, Gnansounou *et al.* 2009, Johnson 2009, Maness 2009, Searchinger *et al.* 2009, Tilman *et al.* 2009, O'Laughlin 2010, Singh *et al.* 2010, Cherubini *et al.* 2011, McKechnie *et al.* 2011, Frieden *et al.* 2012, Schulze *et al.* 2012, Smith and Searchinger 2012, Wibe 2012, Wiloso *et al.* 2012, Zanchi *et al.* 2012, Agostini *et al.* 2013).

A survey of the recent literature suggests several key reasons for these conflicting results, which could significantly influence the greenhouse gas performance of biofuels:

1) consideration of initial carbon stocks (both in above-ground vegetation and in below-ground organic soil) and the potential loss of carbon from vegetation and soil due to direct initial land-use change activities,
2) potential loss of soil carbon in harvest years due to the impact of mechanized activities on soil compaction and scarification,
3) time-dependency of the GHG and carbon emissions and carbon sequestration dynamics during the life of a biofuel project,

4) between-project variability of project-specific inputs and conditions.

Whitaker *et al.* (2010) recently reviewed the GHG balance results of 44 studies examining bioethanol and concluded that it is essential that areas of uncertainty such as soil carbon pools and fluxes and soil GHG emissions be included in GHG balance assessments, and that further research needs to be conducted to enable robust calculations of impacts under different land-use change scenarios. Without the inclusion of these parameters, the authors suggest that it is uncertain that biofuels are really delivering GHG savings compared with fossil fuels.

The importance of the potential impacts of initial carbon stocks has been recognized (Fargione *et al.* 2008, Cherubini *et al.* 2009, Dale *et al.* 2010, Cherubini 2010, Djomo *et al.* 2011) and the greenhouse-gas benefits of biofuels use can be reduced or even become negative if carbon is released and the GHG emissions arising from the associated change of land use are significant (IEA 2010). The European Commission has adopted the Renewable Energy Directive 2003 (European Council 2009) in order to avoid undesirable direct land-use change for the expansion of the biofuel feedstock production area.

There are two sources of potential emissions: loss of carbon from vegetation that is removed by clearing the land and preparing for planting, and loss of long-stored carbon in the soil through tillage operations. The land proposed for plantation or afforestation has an existing (baseline) stock of organic matter and biophysical GHG flux dynamics. If forest or grassland is converted to grow biofuels on said land, the vegetation needs to be cleared, which mostly decomposes or burns, and transfers carbon back to the atmosphere. Also, uncultivated land typically requires ploughing as part of the site planting preparation activities (Adler *et al.* 2007, Berguson 2010, McAuliffe 2011) to make way for the energy crop. This allows oxygen and microbes to break down much of the carbon long stored in the soil, releasing it back into the atmosphere in the form of carbon dioxide, potentially resulting in the biofuel operation having a higher GHG burden than fossil-derived fuel (Wicke *et al.* 2008, Stephenson *et al.* 2010).

The potential loss of carbon from soil due to direct initial land-use change has been identified as a key factor that can significantly influence the greenhouse gas performance of biofuels. Brandao *et al.* (2011) note that many GHG balance studies of biofuels ignore the changes in soil organic carbon associated with growing biomass, outlining the importance of monitoring soil carbon stocks not only at the initial direct land-use change, but also throughout the planning horizon. Cowie *et al.* (2006) notes that initial soil carbon content has a major influence, and the equilibrium soil C stock may be lower than that of the previous pasture. Intensively managed biofuel systems, such as perennial grasses and short rotation woody crops, are likely to have lower equilibrium soil carbon due to more frequent site disturbance and high rate of biomass removal (Cowie *et al.* 2006).

Converting agricultural or marginal land to tree plantations does not necessarily result in a benefit in terms of the terrestrial carbon stocks. Paul *et al.* (2003) suggest that soil organic carbon decreased at an average rate of 1.7 Mg C ha^{-1} yr^{-1} during the first 10 years following afforestation with trees including hardwoods (eucalypt). Findings from other studies also suggest that soils with high initial soil organic carbon contents generally showed losses in carbon immediately (5-10 years) following afforestation (Vesterdal *et al.* 2002, Paul *et al.* 2002).

Laganiere *et al.* (2010) found that studies have reported contradictory findings: afforestation resulted in either a decrease or an increase in soil organic carbon stocks, or had a negligible effect. Nevertheless, a trend appears to emerge: afforestation frequently shows an initial loss in soil organic carbon during the first few years, followed by a gradual return of C stocks to levels comparable to those in the control agricultural soil, and then increasing to generate net C gains in some cases. This finding is especially important in the context of short rotation plantations establishment, where the time between successive harvests and site preparation activities is short.

Two recent reports on soil carbon dynamics studied hybrid poplar plantation establishment on marginal or former agricultural land, monitored up to 20 and 15 years of age, respectively (Mao *et al.* 2010, Zhang *et al.* 2010). In both studies, organic carbon stocks in the soil decreased in the first 10 years (8, respectively) following afforestation.

The impact of mechanized activities on soil carbon losses has been identified as another key factor in biofuels GHG balances (Larson 2006, Whitaker *et al.* 2010). During the subsequent cycles of energy crops operations (*i.e.* at harvest-regeneration time), it has been reported that during tillage operations (e.g. plowing, sub-soiling and harrowing) soil aggregates can be disrupted leading to an accelerated mineralization of soil organic matter (Fernandes *et al.* 1997). Some of the biomass production contained in living plant material is eventually transferred to dead organic matter pools (*i.e.* dead wood, foliage, litter), while other organic matter decomposes quickly, returning carbon to the atmosphere, but a portion is retained for months to years to decades. Land use and management influence carbon stocks of dead organic matter by affecting the decomposition rates and input of fresh detritus. Furthermore, with the increase in green systems, which includes the use of mechanical harvesters, soil compaction is a possibility. Soil compaction may increase N_2O (a major greenhouse gas) emissions due to decreased aeration and stimulation of anaerobiosis and denitrification (Kelliher *et al.* 2003, Bessou *et al.* 2010). Nave at al. (2010) reported losses in mineral soil C following harvest and in surface mineral soil C when tillage is used following harvest.

Recent studies report that the emission benefits of biofuel compared to the use of fossil fuel are time-dependent (Zanchi *et al.* 2012, McKechnie *et al.* 2011, Schlamadinger and Marland 1996), a concept sometimes referred to as carbon debt and payback time. For example, Zanchi *et al.* (2012) used a time-dependent GHG accounting method, however, considered only the biomass consumption emissions from a portion of the harvested biomass, namely the fraction that is used (burned) in the final use stage, and did not consider production chain emissions (e.g., from biomass production activities, biomass transport, biorefinery production emissions). McKehnie *et al.* (2011) showed that benefits of biofuels *vs*. fossil fuels are time-dependent, however, the authors:

- Did not include direct land-use change impacts or emissions from enzyme inputs even though the potential contribution to ethanol life cycle emissions were acknowledged; see also MacLean and Spatari (2009),
- Assumed biomass to be carbon neutral,
- Used GREET model-based data for hybrid poplar dedicated plantations (farmed trees) although the study focused on the GLSL forest region in Ontario which is managed for optimal production of commercial species other than poplar trees, and therefore would not be harvested at an optimal rotation for poplar, or with salvage logging methods not necessarily appropriate for poplar.

In addition, recent reviews suggest that one of the key reasons why the results of biofuel GHG balance analyses are conflicting is that many of the important inputs and emissions are project-specific and have a large between-project variability (von Blottnitz and Curran 2007, Cherubini *et al.* 2009, Singh *et al.* 2010, Whitaker *et al.* 2010, Borrion *et al.* 2012, Wiloso *et al.* 2012, Agostini *et al.* 2013). Methodologies for calculating biofuels GHG balance that simply average out emissions or credits (e.g., sector-level energy input-output models, life cycle analyses that are done for a product at an entire industrial or economic sector level instead of at a project level), or that make generalization assumptions (e.g., biomass emissions are simply assumed carbon neutral), are by design in such a way that they can not properly capture this between-project variability.

Notwithstanding the reported importance of these key areas (*i.e.* initial carbon stocks in vegetation and soil and potential losses of carbon due to direct initial land-use change; potential loss of soil carbon in harvest years due to the impact of mechanized activities; time-dependency of GHG and carbon emissions dynamics; and, between-project variability of project-specific inputs and conditions) they have not been simultaneously and explicitly all included in bioethanol GHG balance analyses.

For example, energy- or material-balance models for GHG analysis of biofuels, some initially developed for annual agricultural feedstocks, such as GREET (Wang *et al.* 2007), GHGenius ((S&T)2 Consultants Inc. 2011), BESS (Liska *et al.* 2009), EBAMM (Farrell *et al.* 2006), SimaPro (PRé Consultants 2012), and ecoinvent (Frischknecht *et al.* 2005): Do not consider direct land-use changes on above- and below-ground carbon stocks, do not include dynamics of biogenic carbon stocks other than harvested biomass, which is simply assumed to be carbon-neutral, and do not consider the timing of carbon emissions and sequestration during the planning horizon.

von Blottnitz and Curran (2007) reviewed the literature on GHG balance of lignocellulosic biofuels for agricultural and waste feedstocks between 1996 and 2004, and reported both favourable and unfavourable GHG balance comparisons between bio-ethanol and the fossil fuel displaced. The authors found only one study that considered soil carbon, with corn stover as feedstock, which assumed no-till practices. von Blottnitz and Curran (2007) did not report on any study that considered carbon impacts from land-use change, soil carbon impacts, or time-dependent effects.

The literature of GHG balance assessments of lignocellulosic ethanol production published between 2005 and 2011 was recently reviewed by Borrion *et al.* (2012) who concluded that the complexity of biofuel systems generates significantly different GHG emissions results due to the differences in input data between projects, methodologies applied, and local geographical conditions. Some of the studies reviewed concluded that there is not a reduction of GHG emission when using lignocellulosic ethanol in comparison to fossil fuel systems. None of the studies reviewed were reported to have considered soil carbon dynamics, carbon impacts from land-use change, and time-dependent effects.

This thesis chapter contributes to the discussion by proposing the Carbon Balance and Biomass to Biofuel Optimization planning model (C3BO), a model for calculating biofuel projects GHG balances that considers all four key areas described above, thus contributing to a better understanding of the factors affecting the viability of displacing fossil fuels with wood ethanol derived from dedicated energy plantations.

The C3BO model incorporates the optimal biomass production strategies determined by the Carbon-aware Biomass production Optimization System (C-BOS) developed in CHAPTER 2, and the Biogenic Carbon Dynamics model (Bio-CarbD) developed in CHAPTER 3. The LP optimization model C-BOS model solves the industrial economic problem (cost minimization) and the Bio-CarbD model calculates the net biogenic carbon balance. One of the added benefits of integrating C3BO with the C-BOS optimization model is the ability to evaluate the impact of various model inputs on the total production costs of the biomass-to-biofuel project, from land-use change and establishing the fast growing tree plantations to the conversion of biomass to ethanol in the biorefinery. As a result, the project costs are presented together with the project GHG balance, which offers an indication of the financial viability of the biofuel project.

4.3 Methodology

The greenhouse gas balance analysis of a wood to ethanol biofuel system is carried out within the C3BO model following the ISO standards on attributional life cycle assessment (ISO 2006, ISO 2006), including guidelines and recommendations suggested by Cherubini

(2010). The goal of the life cycle analysis in the C3BO model is to determine the viability of a biofuel project to displace gasoline with wood ethanol, from a GHG balance perspective. The scope of the life cycle analysis is to account for all the GHG emissions and credits that are caused by the existence of the biofuel project (*i.e.* which would not have occurred in the absence of the project). The comprehensive approach used in the C3BO model is illustrated by the biofuel production activities and processes included within the system boundary, as shown in Figure 4.1.

The functional unit of the life cycle analysis used in C3BO is the quantity of CO_2-equivalent GHG emissions per unit of ethanol energy content that are produced as a result of the biofuel project, expressed as kg CO_2/GJ EtOH (equivalent to g CO_2/MJ EtOH). This provides a reference to which all other emissions and credits can be related, including the emissions of the displaced gasoline system.

The viability of displacing gasoline with wood ethanol (from a GHG balance perspective) is determined in the C3BO model by:

1. Quantifying the net GHG-equivalent emissions over the planning horizon produced as a result of the existence of the biofuel project (including any credits for net carbon sequestration),
2. Determining the emissions associated with a fossil fuel system that would produce an equivalent quantity of transportation fuel,
3. Comparing the GHG balances of the two systems to determine whether the biofuel production project produces less or more emissions than the displaced fossil fuel system.

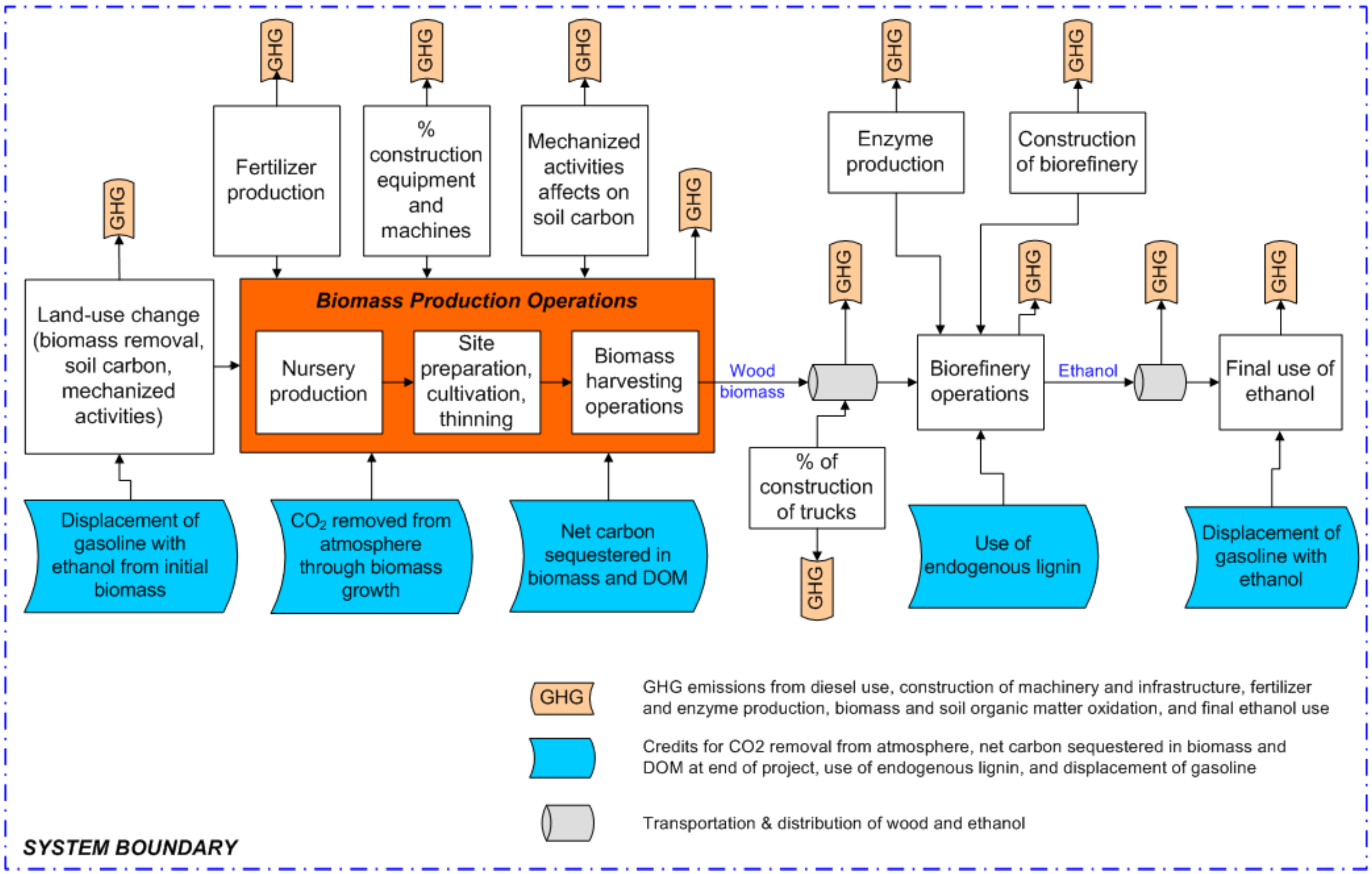

Figure 4.1. Biofuel production system boundary.

4.3.1 Net GHG emissions of the biofuel project compared with the equivalent fossil fuel production system

As mentioned earlier, the C3BO model builds on the harvest-regeneration model C-BOS and the biogenic carbon balance and analysis model Bio-CarbD. Solving the C-BOS model optimization problem results in a schedule of production activities for each land unit, including which treatment sequences will be applied to which land parcels over the duration of the project, and the quantity of biomass (with its respective carbon content) contained in each carbon pool in each year. This information is used *a posteriori* in Bio-CarbD to calculate the carbon stocks and fluxes for the biofuel project.

In addition to the biogenic carbon removals and emissions by live biomass and DOM (discussed earlier in the Bio-CarbD model in CHAPTER 3), the overall project carbon and GHG balance model C3BO considers additional life-cycle GHG emissions from use of fossil fuels (diesel) by forest operations machinery and biomass transportation trucks, and conversion to biofuel operations at the biorefinery.

The carbon and GHG equivalent emissions are calculated for three key greenhouse gases (IPCC 2006): carbon dioxide (CO_2), methane (CH_4), and nitrous dioxide (N_2O). The diesel emission factors for CH_4 and N_2O are expressed as CO_2 -equivalents by using their Global Warming Potential (GWP) factors on a 100-year time horizon basis (IPCC 2007). The 100-year basis can be adjusted to correspond to the project horizon. For shorter projects, a 20-year basis could be used instead.

To obtain the overall net life-cycle carbon and GHG equivalent emissions balance of the biofuel project compared with the equivalent fossil fuel production system, the total GHG emissions balance of the biofuel project is summed with the credits for the displacement of fossil fuel emissions (equation 4.1).

$$netGHG\_PROJ_{em} = GHG\_PROJ_{em} + FOSS_{cr} \qquad 4.1$$

where,

$netGHG\_PROJ_{em}$ = the net life-cycle GHG emissions of the biofuel project, when accounting for the credits from the emissions of the displaced fuel (kg CO_2/GJ EtOH, equivalent to g CO_2/MJ EtOH).

GHG_PROJ_{em} = total GHG emissions balance of the biofuel project. This is further explained in section 4.3.1.1 below (kg CO_2/GJ EtOH).

$FOSS_{cr}$ = total emissions credit for the fossil fuel that is being displaced by the ethanol produced in the project; this is sometimes called the fuel-switch credit. This is further explained in section 4.3.1.2 below (kg CO_2/GJ EtOH).

4.3.1.1 Life-cycle GHG emissions balance of the biofuel project

The total GHG emissions balance of the biofuel project is calculated as the sum of total emissions from the production of biomass and conversion to biofuels including biomass transport, and the credit for net carbon sequestered on site at the end of planning horizon, in live biomass and in DOM pools (equation 4.2).

$$GHG\_PROJ_{em} = GHG\_PROD_{em} + CSEQ_{cr} \qquad 4.2$$

where,

GHG_PROD_{em} = total emissions from the production of biomass and conversion to biofuels, including biomass transport (kg CO_2/GJ EtOH).

$CSEQ_{cr}$ = credit for net carbon sequestered on site at the end of planning horizon, in live biomass and in DOM pools; being a credit, this quantity has a negative sign, by convention in this study (kg CO_2/GJ EtOH).

The total emissions from the production of biomass and conversion to biofuels, including biomass transport, GHG_PROD_{em}, is calculated using equation 4.3.

$$GHG\_PROD_{em} = GHG\_iLUC_{em} + GHG\_iMEC_{em} + GHG\_pMEC_{em} + GHG\_iTRN_{em} + GHG\_pTRN_{em} + GHG\_iINFR_{em} + GHG\_iREF_{em} + GHG\_pREF_{em} \qquad 4.3$$

where,

GHG_iLUC_{em} = GHG emissions from removal of the pre-existing biomass. These emissions include biogenic losses of carbon from any biomass that is removed at land-use change when the land units are prepared for planting. This biomass is assumed to be converted to ethanol. An appropriate credit is given for displacing the equivalent quantity of gasoline ($iFOS_{cr}$ explained in section 4.3.1.2 below) and for utilizing some of the lignin recovered from the conversion process ($iLIG_{cr}$ explained below). (kg CO_2/GJ EtOH).

GHG_iMEC_{em} = GHG emissions from initial mechanized harvesting-processing activities of the initial pre-existing biomass: felling; skidding; delimbing; loading; mobile chipping on site or debarking/chipping at biofuel conversion facility; this includes the embodied emissions from construction of machinery, as a proportion of machinery use from machinery lifetime (kg CO_2/GJ EtOH).

GHG_pMEC_{em} = GHG emissions from project mechanized biomass production activities of the biofuel project (kg CO_2/GJ EtOH).

GHG_iTRN_{em} = GHG emissions from transport of initial pre-existing biomass to the biofuel conversion facility with logging trucks (for transport of stems) or chip trucks (if the harvested biomass is chipped at the forest site); this includes the embodied emissions from construction of trucks, as a proportion of truck use from truck lifetime (kg CO_2/GJ EtOH).

GHG_pTRN_{em} = GHG emissions from transport of project biomass to the biofuel conversion facility with logging trucks (for transport of stems) or chip trucks (if the harvested biomass is chipped at the forest site) (kg CO_2/GJ EtOH).

GHG_iINFR_{em} = GHG emissions from initial construction of biorefinery facility infrastructure (kg CO_2/GJ EtOH).

GHG_iREF_{em} = GHG emissions from biorefinery activities to convert initial pre-existing biomass into biofuel product (kg CO_2/GJ EtOH).

GHG_pREF_{em} = GHG emissions from biorefinery activities to convert project biomass into biofuel product (kg CO_2/GJ EtOH).

The GHG emissions from project mechanized biomass production activities, GHG_pMEC_{em}, is calculated using equation 4.4.

$$GHG\_pMEC_{em} = GHG\_pPRP_{em} + GHG\_pSLV_{em} + GHG\_pHRV_{em} \quad 4.4$$

where,

GHG_pPRP_{em} = GHG emissions from site preparation activities: mechanized herbicide application; deep plow to 30 cm deep; ripping; pre-emergent herbicide (kg CO_2/GJ EtOH).

GHG_pSLV_{em} = GHG emissions from mechanized silviculture-management activities: pre-plant herbicide; cultivation; herbicide weed control; fertilization (including emissions from fertilizer production) (kg CO_2/GJ EtOH).

GHG_pHRV_{em} = GHG emissions from mechanized harvesting-processing activities of the project biomass: felling; skidding; delimbing; loading; mobile chipping on site or debarking/chipping at biofuel conversion facility; this includes the embodied emissions from construction of machinery, as a proportion of machinery use from machinery lifetime (kg CO_2/GJ EtOH).

The GHG emissions from biorefinery activities to convert initial pre-existing biomass into biofuel product, GHG_iREF_{em}, are calculated with equation 4.5.

$$GHG\_iREF_{em} = GHG\_iCNV_{em} + iLIG_{cr} \quad 4.5$$

where,

GHG_iCNV_{em} = GHG emissions from industrial processes of converting the initial pre-existing biomass to biofuel at the biorefinery, and from distribution of biofuel product to blending stations. The emission factors for the conversion facility are assumed from data available in GHGenius ((S&T)2 Consultants Inc. 2010, (S&T)2 Consultants Inc. 2010) and other references (Jungbluth *et al.* 2007, U.S. Department of Energy 2010). It is important that the emissions from distribution of the biofuel product are representative of a system that is similar to the fossil fuel distribution system (*i.e.* emissions from production and distribution of the displaced fossil fuel are considered up to the same blending stations), to ensure that the emissions comparison is consistent (kg CO_2/GJ EtOH).

$iLIG_{cr}$ = credit given for utilizing some of the lignin recovered from the conversion process (for processing the initial pre-existing biomass) in order to produce energy that is input back into the conversion process, if this displaces a fossil fuel product (kg CO_2/GJ EtOH).

The GHG emissions from biorefinery activities to convert project biomass into biofuel product, GHG_pREF_{em}, is calculated using equation 4.6.

$$GHG\_pREF_{em} = GHG\_pCNV_{em} + pLIG_{cr} \tag{4.6}$$

where,

GHG_pCNV_{em} = GHG emissions from industrial processes of converting the project biomass to biofuel at the biorefinery, and from distribution of biofuel product to blending stations (kg CO_2/GJ EtOH).

$pLIG_{cr}$ = credit given for utilizing some of the lignin recovered from the conversion process (for processing the project biomass) in order to produce energy that is input back into the conversion process, if this displaces a fossil fuel product (kg CO_2/GJ EtOH).

The credit for net carbon sequestered on site at the end of planning horizon in live biomass and in DOM pools, $CSEQ_{cr}$, is calculated using equation 4.7.

$$CSEQ_{cr} = CATM_{cr} + iDOM + GHG\_pBMH_{em} + GHG\_DCAY_{em} - iDOM \tag{4.7}$$

where,

$CATM_{cr}$ = credit for the net removal of carbon from the atmosphere, which includes both the removal of carbon through biomass growth and the emissions from harvest and DOM decay, as described earlier in the Bio-CarbD model (kg CO_2/GJ EtOH).

iDOM = carbon stock in DOM pools after land-use change. Besides the carbon from initial DOM before land-use change, this includes the transfers from foliage, coarse roots and fine roots to the DOM pools as a result of the land-use change. In absolute terms, this is the same quantity as -iDOM, and mathematically they cancel each other out. This quantity is introduced in the model in order to be able to track the changes to the initial DOM pools through time, and it is mentioned here to help explain the approach. The negative sign (-

iDOM) also represents an emissions credit that accounts for the fact that the initial DOM pools would have contained a constant amount of carbon throughout the years in the absence of the biofuel project (kg CO_2/GJ EtOH).

GHG_pBMH_{em} = GHG emissions from harvested biomass generated by the project (not including initial pre-existing biomass, which is part of the GHG_iLUC_{em} calculation). This represents the biomass removed from the plantation through harvesting, and includes the removed stem, bark, and branch pools. During one year of the biofuel project, most of the harvested stem, bark and branch biomass ends up in the biofuel product; however, the remaining portion of the harvested biomass that does not end up in the biofuel is assumed to be oxidized (*i.e.* its carbon released into atmosphere) during the same year through losses during processing and conversion (kg CO_2/GJ EtOH).

GHG_DCAY_{em} = GHG emissions from natural decay of matter in DOM pools (kg CO_2/GJ EtOH).

It is important to note that the emissions from the final combustion of the biofuel are included in the carbon and GHG balance calculations as part of the emissions in GHG_pBMH_{em}; this ensures avoiding double counting of the emissions resulted from the harvested biomass.

The carbon stock of the DOM pools immediately following land-use change, iDOM, is calculated using equation 4.8.

$$iDOM = iDDM + iDFF + iDCR + iDFR \qquad 4.8$$

where,

iDDM = initial carbon stock in pre-existing DOM pools (*i.e.* quantity of initial dead organic matter before land-use change) (kg CO_2/GJ EtOH).

iDFF = initial carbon stock in DOM pools resulting from transfer from the foliage of pre-existing biomass, at land-use change (kg CO_2/GJ EtOH).

iDCR = initial carbon stock in DOM pools resulting from transfer from the coarse roots of pre-existing biomass, at land-use change (kg CO_2/GJ EtOH).

iDFR = initial carbon stock in DOM pools resulting from transfer from the fine roots of pre-existing biomass, at land-use change (kg CO_2/GJ EtOH).

4.3.1.2 Emissions credit for the fossil fuel that is being displaced by the biofuel produced in the biofuel project

The emissions credit for displacing fossil fuel with the ethanol produced in the biofuel project is calculated with equation 4.9:

$$FOSS_{cr} = iFOS_{cr} + pFOS_{cr} \tag{4.9}$$

where,

$iFOS_{cr}$ = emissions credit for displaced fossil fuel at initial land-use change: if the biomass removed from the site at land-use change is converted to a biofuel product, then the portion of the carbon in the respective biomass which is reconstituted in the biofuel product is considered a credit if it replaces a fossil-derived fuel (kg CO_2/GJ EtOH).

$pFOS_{cr}$ = emissions credit for displaced fossil fuel during the project, from using the biofuel resulted from the trees planted, specifically for the biofuel project (kg CO_2/GJ EtOH).

The Bio-CarbD model converts the CO_2-equivalent emissions resulting from biomass production to carbon-equivalents, expressed on a per hectare per year basis. This calculation is possible since the available production activities for all treatments are known for each land parcel, as well as the rotation age and the growth and yield.

4.3.2 GHG savings of biofuel project

As a measure of the climate change mitigation potential of the biofuel project, the C3BO model considers the life-cycle GHG emission reductions of the biofuels production system compared with a reference system for the emissions of the displaced petroleum-based fuels system, over a time horizon. The baseline represents the likely conditions in the absence of the biofuel project, both in the biomass production and land use and in the conversion to biofuel aspects.

In terms of the baseline for biomass production and land use, the absence of the biofuel project is assumed to mean that the land areas, which would be used for the biofuel project, remain idle (*i.e.* unused for any commercial or managed operations) under a natural regime. To simplify the carbon balance comparisons with the reference system it is assumed that in the baseline condition the carbon balance of the land areas would be neutral, *i.e.* at equilibrium. Although the formulation of our model would permit the comparison with any other reference system of activities. This means that the amount of carbon sequestered by the live biomass and soil during the time horizon is equivalent to the amount of carbon that is released back into the atmosphere through natural decay of dead organic matter and respiration of biomass and soils.

If any pre-existing biomass were to be removed from the production areas at the beginning of the project (*i.e.* at land-use change) in order to facilitate the planting of trees, then the carbon content of this biomass would be subtracted from the carbon balance of the biofuel project.

In terms of the baseline for conversion of biomass into biofuel and final use of biofuels, the absence of the biofuel project is assumed to mean that a certain quantity of petroleum-based fuel will be used during the time horizon. The quantity of these fossil fuels is equivalent (on an energy content per unit of volume basis) to the quantity of biofuels produced by the biofuel project. The displacement of fossil fuels with biofuels would be considered as a credit (sometimes called "fuel switch credit") in the GHG balance of the biofuel project.

The project time horizon, also called the planning period, can be anywhere from a few years to several decades into the future. The C3BO model formulation permits for any number of years to be input as the time horizon for the analysis.

The general equation for calculation of GHG emissions reduction of the biofuel project, reported as a proportion of the displaced baseline fossil fuel $FOSS_{cr}$ (in a similar fashion as in US EISA and RFS regulations, (U.S. EPA 2010, Yacobucci and Bracmort 2010)), is shown in equation 4.10:

$$GHG\_SAVINGS = \frac{-FOSS_{cr} - GHG\_PROJ_{em}}{-FOSS_{cr}} \qquad 4.10$$

The negative sign applied to $FOSS_{cr}$ is meant to bring the respective value back to a positive number, since $FOSS_{cr}$ is an emissions credit and is considered a negative number by convention in this study.

4.3.3 Biofuel production financial model

As mentioned earlier, one of the added benefits of integrating the C3BO model with the C-BOS optimization model is the ability to evaluate the impact of various model inputs on the total production costs of the biomass-to-biofuel project, from land-use change and establishing the fast growing tree plantations to the conversion of biomass to ethanol in a biorefinery. The biofuel production financial model consists of two key components: 1) the operating costs of biomass production and transport to the biorefinery, and 2) the capital and operating costs of the biorefinery.

As described in CHAPTER 2 the delivered biomass production costs (*i.e.* the costs "at the biorefinery gate") represent the costs associated with: site preparation including removal and processing of any pre-existing biomass at land-use change; land rent; seedlings and planting; silviculture, irrigation and fertilization; harvesting and biomass processing at the forest site; and, biomass transportation to the biorefinery.

The capital and operating costs of the biorefinery represents the annual and per unit costs of biofuel production, such as chemical components and mixtures, equipment sizing, and economic-evaluation parameters such as financing, depreciation, running royalty expenses, inflation rate and taxes – depending on the conversion technology used and the size of the biorefinery.

The total cost of producing the biofuel product is calculated with equation 4.11:

$$TOTAL\_COST = iCOST + iREFI + pCOST + pREFI \qquad 4.11$$

where,

TOTAL_COST = the total cost of producing the biofuel product, including: harvesting, transporting and converting the initial pre-existing biomass at land-use change; preparing the land that will be used to grow the project biomass; land rent for the land units that are used in the biofuel project; project biomass production activities; transporting the project biomass to the biofuel facility; and, conversion of project biomass into biofuel product [$/unit volume of biofuel product].

iCOST = production cost of processing the initial pre-existing biomass at land-use change [$/unit volume of biofuel product].

iREFI = cost of converting the initial pre-existing biomass into a biofuel product at a biofuel facility, both capital and operating [$/unit volume of biofuel product].

pCOST = project cost of biomass production. This is calculated by the optimization model C-BOS using the expression shown in equation 2.1 [$/unit volume of biofuel product].

pREFI = costs of converting the project biomass into a biofuel product at a biofuel facility, both capital and operating [$/unit volume of biofuel product].

The production cost of processing the initial pre-existing biomass at land-use change is calculated with equation 4.12:

$$iCOST = iPROD + iTRAN \tag{4.12}$$

where,

iPROD = costs of harvesting the initial pre-existing biomass at land-use change [$/unit volume of biofuel product].

iTRAN = costs of transporting the initial pre-existing biomass to the biofuel facility, at land-use change [$/unit volume of biofuel product].

The project cost of biomass production is shown in equation 4.13:

$$pCOST = pPREP + pRENT + pPROD + pTRAN \tag{4.13}$$

where,

pPREP = similar to the expression *PREP* described above in section 2.3.1, expressed per unit quantity of the biofuel product [$/unit volume of biofuel product].

pRENT = similar to the expression *RENT* described above in section 2.3.1 [$/unit volume of biofuel product].

pPROD = similar to the expression *PROD* described above in section 2.3.1 [$/unit volume of biofuel product].

pTRAN = similar to the expression *TRAN* described above in section 2.3.1 [$/unit volume of biofuel product].

4.3.4 The structure of the C3BO model

The C3BO model combines a multi-period, multi-area biomass production optimization model with a detailed carbon dynamics and balance model, to analyze the financial-economic and environmental costs and benefits of using forest biomass to generate biofuel. Overall the C3BO model includes four key components:

1. A biomass production planning model: the carbon-aware biomass production optimization system (C-BOS), which determines the scheduling and costing of biomass production activities for a biofuel project over a long-term planning horizon (*i.e.* 30-100 years), including site preparation, planting and silviculture, biomass production, harvesting and processing, and transportation to the ethanol production facility,
2. A carbon balance model for the biomass production system: the biogenic[4] carbon dynamics model (Bio-CarbD) with a yearly time step, which includes carbon removals from the atmosphere through biomass growth, carbon sequestration in live biomass and DOM pools, and carbon emissions from DOM decay and biomass removals through harvest,

[4] The term "biogenic carbon" in this dissertation refers to the organic nature of biomass. Similarly, biogenic carbon emissions refer to emissions from biodegradation (natural decay) or combustion of organic material. In contrast, non-biogenic carbon emissions result from the combustion of fossil-fuel based products.

3. A biofuel project-level financial-economic module, which includes the production costs of biomass production activities (item 1 above), as well as the activities of biomass conversion into biofuel,
4. A biofuel project-level carbon and greenhouse gas balance module, which includes emissions from all activities in biomass production, transport, conversion to biofuel, and emission credits from using bark and lignin in the biofuel conversion process to displace a fossil fuel.

The component modules and their overall integration in the C3BO framework is presented in Figure 4.2.

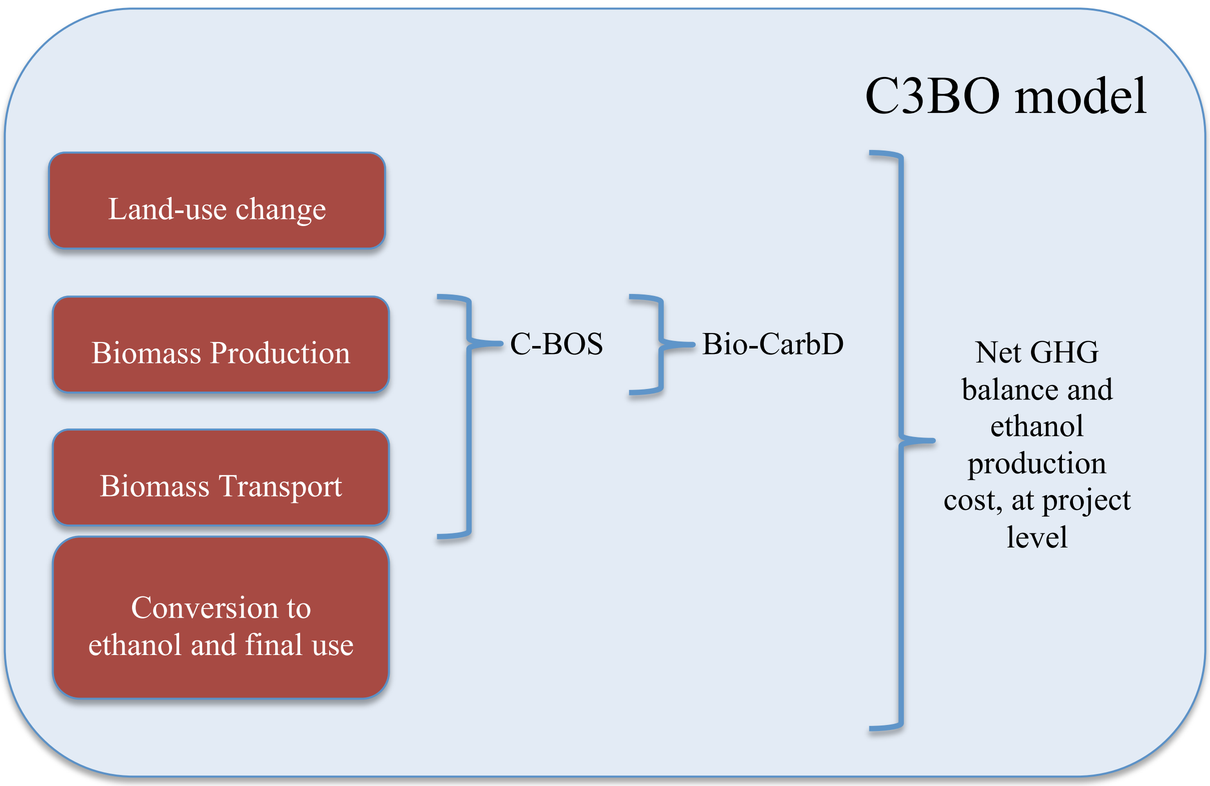

Figure 4.2. C3BO model components

This is a general overview of how the C3BO model is structured:

1. The C-BOS optimization model creates a biomass production plan over the project time horizon, including the scheduling of planting and harvesting activities within each land parcel, and the associated production costs,
2. The biomass growth information, as well as the harvesting and planting activities sequence determined by C-BOS, are then used within Bio-CarbD to calculate the dynamics of biogenic carbon within each land parcel by individual carbon pools, including the sequestration of CO2 from the atmosphere through biomass growth, and the emissions of CO2 through decay from DOM pools,
3. The project-level production costs module calculates the project-wide biofuel production costs: it includes the biomass production costs generated by C-BOS, as well as the costs of converting the biomass into ethanol at the biorefinery,
4. The project-level carbon and GHG balance module includes the biogenic carbon dynamics from production of biomass generated by Bio-CarbD, the carbon emissions from land-use change, as well as all the GHG emissions associated with mechanized activities for land-use change, biomass production, biomass transport, conversion to biofuel, and final use of the biofuel.

4.4 Test case for a prototype biofuel production system

As an illustration and practical application of the proposed Carbon Balance and Biomass to Biofuel Optimization planning system (C3BO), we present a test case for a biofuel project producing ethanol in a biorefinery using biomass from an afforestation plantation with fast growing poplar trees.

Some of the key assumptions about the biomass production activities and costs, ethanol conversion factors, biorefinery techno-economic assumptions, as well as the biogenic carbon dynamics and accounting, have been presented in CHAPTER 2 and CHAPTER 3. In this section we present the key assumptions in our analysis for the following elements: land units; transportation of biomass; biorefinery techno-economic assumptions; emissions from biomass production and conversion activities; planning horizon; the baseline case (*i.e.* business as usual; absence of the biofuel project); and, the test case scenarios.

4.4.1 Land units

For simplicity and demonstration of the model structure, we considered only two land unit types for the test case: one irrigated and one non-irrigated. On the irrigated land unit the possible treatment types were single stem irrigated (Si) and coppice irrigated (Ci). For the non-irrigated land unit, the treatment types available were single stem non-irrigated (S) and coppice non-irrigated (C).

Each land unit type was assumed to be sufficiently large (*i.e.* unconstrained number of hectares) as to accommodate any planting area size that would be needed to produce the necessary quantity of biomass. The model has the capability, however, to model multiple areas, each with its own defined land size.

The biomass production strategies (treatments), activities and costs have been explained in CHAPTER 2. The carbon content of biomass material was calculated using a carbon conversion factor of 0.5 (Penman *et al.* 2003).

4.4.2 Transportation of biomass to biorefinery

The transportation distances between land units and biorefinery can be specified in the C-BOS model for each land unit type separately. For the test case the transportation distance was assumed to be 40 km (low cost scenarios) or 200 km (high cost scenarios). This range is larger than the 64-112 km assumed by National Research Council (2009), and it was selected in order to investigate the effects of lower and higher transportation distances.

The transportation of harvested material from land units to the biorefinery is assumed to employ log trucks (for transporting tree stems, which will be further chipped at the biorefinery) and chip trucks (for transporting wood chips that have been comminuted on the land units at harvest time). The harvested stems from the single stem production treatments (S and Si) and from the non-irrigated coppice (C) were assumed to be transported by log trucks. The wood chips from the irrigated coppice (Ci) treatments was assumed to be transported by chip trucks.

The costs of transporting the harvested biomass (logs or chips, depending on the production method of each land parcel) to the biorefinery were calculated on a per-unit and per-kilometre basis using costing parameters referenced in the literature (McKechnie *et al.* 2011, Zhang *et al.* 2010, Gautam *et al.* 2010, Sambo 2002, Sessions 2010). Using a transportation distance of 40 km, the roads were assumed to be: dirt 2 km, gravel 5 km, and paved 33 km, with respective i) average travel speeds of 4.8 km/h, 24.1 km/h and 88.5 km/h; and ii) average fuel consumption rates of 60 l/h, 45 l/h, and 30 l/h (Sessions 2010). The average payload was assumed to be 18.75 dry tonnes, and the average hourly rate was $127/h. The average transport unit cost for logs using log trucks was calculated to be 0.74 $/tonne/km.

The transport cost for chips using chips trucks was calculated using an hourly rate of transportation of $85/h and a load weight of 15.88 dry tonnes/load (Gautam *et al.* 2010). For a 40 km transportation distance the transport cost for chip trucks was calculated to be 0.52 $/tonne/km.

The reference cost for diesel fuel used in the transportation cost analysis was assumed to be $1.20/l.

The ethanol conversion factors and biorefinery production capacity have been explained in CHAPTER 2.

4.4.2.1 Biorefinery production costs

The test case costs for biorefinery production were calculated following the techno-economic model described in the National Research Council of the National Academies report on liquid transportation fuels from biomass (National Research Council 2009). The unit ethanol production costs reported in the National Research Council study included costs for (a) biomass feedstock, (b) enzymes, (c) revenues from surplus electricity sales, as well as (d) capital, operations, and raw materials (other than biomass and enzymes). We adapted the National Research Council costs as follows:

a) The cost of biomass feedstock is not included in the calculation of conversion cost, as it is calculated separately in the C-BOS model.
b) The enzyme (cellulase) cost was calculated to be between $0.11/l EtOH and $0.34/l EtOH for a conversion yield of 234 l EtOH/dry tonne wood, and between $0.03/l EtOH and $0.09/l EtOH for a conversion efficiency of 299 l EtOH/dry tonne wood; using a loading factor of 7.12 kg cellulase/tonnes dry wood, 40.30% cellulose content in wood, and a unit cost of cellulase between $2.48/kg (National Research Council 2009) and $8.00/kg (Klein-Marcuschamer et al. 2012).

c) During the conversion of biomass to ethanol, process electricity is produced from the wood substrate by using a boiler and steam generator. A portion of the produced electricity (15%) was used to power the conversion processes, and the surplus (85%) was assumed to be sold to the grid. The price obtained from selling the excess electricity for the test case was calculated to be between \$0.13/l EtOH and \$0.29/l EtOH, assuming a price for electricity of \$0.05/kWh and a conversion efficiency of wood to surplus electricity of 749 kWh/tonne dry wood feedstock. The calculation assumed 23.70% lignin content in wood on a dry basis, 26.7 kJ/g energy content of lignin, and 50% overall efficiency of converting lignin from dry wood feedstock to electricity (National Research Council 2009).
d) The capital, operations and raw materials cost at the biorefinery (including enzymes cost and excess electricity revenue, but excluding biomass delivered cost) was calculated to be between \$0.29/l EtOH (high capacity 100 mil. gal EtOH/year) and \$0.40/l EtOH (low capacity 40 mil. gal EtOH/year), using the total ethanol production cost referenced in the National Research Council study for the same test case assumptions, subtracting the enzyme costs and adding the electricity revenue. This compares with Viikari *et al.* (2012) who suggest that US studies generally forecast a lower cost of ethanol (USD \$0.34–0.48/l EtOH) than EU studies (USD \$0.57–0.78/l EtOH) (assuming a currency conversion of 1 € ≈ 1.4 USD \$).

4.4.3 Emissions from biomass production and conversion to ethanol activities

As stated earlier, in addition to the carbon removals and emissions by live biomass and DOM (presented earlier in the Bio-CarbD model), the overall project carbon and GHG balance model C3BO considers the life-cycle GHG emissions from use of fossil fuels (diesel) by forest operations machinery and processes, biomass transportation trucks, and conversion to biofuel operations. We also include here a discussion about the carbon emissions from land-use change activities.

4.4.3.1 Emissions from biomass production operations

The biomass production activities that were considered in the analysis of emissions for the test case included: seedlings production and transportation; ripping; pre-emergent, pre-plant and weed control herbicide application; cultivation; fertilizer manufacture and application; irrigation; harvesting, processing and loading. The description of calculations is presented below and the resulting values are shown in Table 4.1 through Table 4.3.

The emissions factors for seedlings production and transportation were calculated using i) an emission factor of 237.54 MJ fuel/1000 seedlings (Kilpelainen *et al.* 2011), ii) an energetic diesel emission factor[5] of 92.28 kg CO_2/GJ of fuel used ((S&T)2 Consultants Inc. 2010) (GHGenius v3.2), and iii) a scaling factor of 0.7[6] to account for smaller unit emissions with larger quantities of seedlings produced and transported.

The emission factors for deep plow (30 cm deep) were calculated assuming the use of i) a tractor 130 MFWD with a productivity of 3.44 ha/hr and diesel fuel consumption of 21.65 l/hr, and ii) a Chisel Plow 15 ft attachment with an associated fuel consumption factor of 5.61 l/ha (Lazarus 2011). Deep plow was applied only once at land-use change for all treatments and planted land parcels.

The emissions factors for ripping 60 cm deep were calculated assuming the use of i) a tractor 160 MFWD with a productivity of 2.50 ha/hr and diesel fuel consumption of 26.65 l/hr, and ii) a V-Ripper 25" O.C. 10 ft attachment with an associated fuel consumption factor of 9.26 l/ha (Lazarus 2011). Ripping was applied only once per rotation for all treatments, in the first year of the rotation.

The emission factors for pre-emergent, pre-plant, and weed control herbicide application were calculated assuming the use of a Boom Sprayer, Self-Prop 60 Ft with a diesel fuel consumption of 1.03 l/ha (Lazarus 2011). The annual emissions factor for the herbicide application operations was calculated at 2.29 kg CO_2/ha/yr for treatment S (single stem), 3.06 kg CO_2/ha/yr for treatment Si (single stem irrigated), 3.18 kg CO_2/ha/yr for treatment C (coppice), 2.69 kg CO_2/ha/yr for treatment Ci (coppice irrigated). The pre-emergent

[5] The energetic diesel emission factor was calculated using a volumetric energy density for diesel of 0.038653 GJ/l diesel and a diesel emission factor of 3.57 kg CO_2/l (includes emissions of 2.663 kg CO_2/l from end use and 0.904 kg CO_2/l from upstream production, (Environment Canada , (S&T)2 Consultants Inc. 2010)

[6] The scaling factor of 0.7 was used in the calculations as follows: first we calculated an emission factor x = 29.53 kg CO_2/ha for treatment S with a = 1347 seedlings/ha, then we calculated the emission factor y for a planting density b using the formula $y = \mathrm{n}x^*(b/a)^\wedge 0.7$

herbicide was applied only once per rotation for all treatments, in the first year of each rotation. The pre-plant herbicide was applied in the first year of each rotation and after each harvest; for the single stem treatments this was done once per rotation, and for the coppice treatments it was also done after each intermediate harvest. The weed control herbicide was applied in years 1-3 for treatment S, years 1-2 for treatment Si, years 1-4, 6-8, 11-13 for treatment C, and in years 1-2, 4, 7, 10, 13 for treatment Ci.

The emission factors for cultivation operations were calculated assuming the use of i) a tractor 130 MFWD with a productivity of 4.17 ha/hr and diesel fuel consumption of 21.65 l/hr, and ii) a Row Cultivator 8 Row-30, 20 ft attachment with an associated fuel consumption factor of 4.12 l/ha (Lazarus 2011). Cultivation was applied in years 1-3 for treatment S, and years 1-3, 6-7, and 11-12 for treatment C. Cultivation was not part of the management operations for the irrigated treatments.

The emission factors for fertilizer manufacture and application were calculated using i) average application rates of 37.50, 58.33, 33.33, and 80.00 kg nitrogen/ha (McAuliffe 2011, Berguson 2010) for treatments S, Si, C, and Ci respectively, ii) emission factors of 0.01 kg N_2O -N/kg N input, and 0.1 kg N_2O -N/ha/yr (IPCC 2006), and iii) a global warming potential for N_2O of 298 for a 100-year time horizon. Fertilizer was applied in years 1, 5 for treatment S, years 1-2, 4, 6 for treatment Si, years 1, 5, 10 for treatment C, and years 1-15 for treatment Ci.

Table 4.1. Emission factors for biomass production activities, excluding harvesting [kg CO_2/ha/yr]

	Treatment			
Emission factors for biomass production activities	S	Si	C	Ci
Seedling production and transport	3.79	5.54	2.47	4.01
Ripping	9.12	12.16	4.86	4.86
Herbicide application	2.36	3.14	3.27	2.76
Cultivation	25.58	--	27.28	--
Fertilizer manufacture and application	0.94	2.51	0.75	3.77

The emission factors for irrigation activities were discussed by Rothausen *et al.* (2011) who reviewed 15 studies on energy use and GHG emissions in irrigation agriculture, and found the average reported emissions from irrigation activities to be 8,529 MJ/ha with a standard deviation of 6,513 MJ/ha. Lal (2004) found that the energy use for irrigation varies widely from roughly 3,000 to 130,000 MJ/ha. Irrigation, lifting water from deep wells and using sprinkling systems, emits 129±98 kg C for applying 25 cm of water and 258±195 for 50 cm of water. Carbon emissions for pump irrigation were estimated at 150–200 kg C/ha/year depending on the source of energy, and for drip irrigation were estimated at 216 kg C/ha/year (Lal 2004).

For the test case we assumed an emission factor from irrigation activities of 37 kg CO_2/ha (low emissions scenarios) and 342 kg CO_2/ha (high emissions scenarios), using an electricity emission factor of 0.044 kg CO_2/kWh, an energetic emission factor of 12,229 g CO_2/GJ ((S&T)2 Consultants Inc. 2010), and an energy content factor for electricity of 0.0036 GJ/kWh (BC MoE 2011).

The emission factors for harvesting operations were calculated separately for the single-stem (S and Si) and for the coppice (C and Ci) treatments. For the single stem treatments (including non-irrigated coppice which was essentially a single-stem production), the component activities were felling, skidding, loading, and processing, including grinding. For the irrigated coppice treatment a modified harvester, trailer tractor and blower were used.

The emission factors for single-stem felling were calculated using a feller buncher with a productivity of 31.50 green tonnes wood/hr and a diesel fuel consumption rate of 25 l/hr (Gautam *et al.* 2010). The emissions factors for single-stem skidding were calculated using a grapple skidder with a productivity of 31.50 green tonnes wood/hr and a diesel fuel consumption rate of 20 l/hr (Gautam *et al.* 2010). The emissions factors are comparable with those reported in Kilpelainen (2011) and Adler (2007). The production factors were applied to the biomass being harvested (stem, bark and branch).

The emission factors for single-stem delimbing were calculated using a pull-through delimber with a productivity of 26.80 green tonnes wood/hr and a diesel fuel consumption rate of 20 l/hr (Hartsough *et al.* 2000). The production factors were applied to the biomass being processed (stem, bark and branch).

The emission factors for mobile chipping of branches at the plantation site (for single stem treatments including coppice non-irrigated) were calculated using a loader with a productivity of 31.70 green tonnes wood/hr and a diesel fuel consumption rate of 15 l/hr (Gautam *et al.* 2010). The production factors were applied to the branch biomass.

The emission factors for single-stem loading were calculated using a loader with a productivity of 31.70 green tonnes wood/hr and a diesel fuel consumption rate of 15 l/hr (Gautam *et al.* 2010). The production factors were applied to the harvested biomass (stem, bark and branch).

The emission factors for on-site (*i.e.* at the biorefinery) chipping of single stems (for single stem treatments including coppice non-irrigated) were calculated using a electric ring style debarker with an energy consumption rate of 8.5 kWh/tonne of raw material, and a chipper and conveyor with an energy consumption rate of 30.3 kWh/tonne of raw material (Martin *et al.* 2000).

Table 4.2. Emission factors for biomass harvesting and processing activities for single stem treatments [kg CO_2/ha/yr]

	Treatment S	Treatment Si	Treatment C		
Rotation			1st	2nd	3rd
Felling	77.59	99.82	77.59	80.69	83.82
Skidding	62.07	79.86	62.07	64.55	67.06
Delimbing	72.95	93.86	72.95	75.87	78.82
Grinding Mobile	53.08	68.56	53.68	55.83	58.06
Loading	46.26	59.52	46.26	48.11	49.98
Grinding Stationary	35.13	45.15	35.01	36.41	37.81

The emission factors for coppice harvesting were calculated using i) a modified harvester with productivity 80 green tonnes/hr and diesel fuel consumption 60 l/hr, ii) a trailer tractor with diesel fuel consumption of 10 l/hr, and iii) a blower with diesel fuel consumption of 5 l/hr (Buchholz and Volk 2011). The increase in emissions from one harvest to the next (Table 4.3) is due to the increased amount of biomass to be harvested (as mentioned above, subsequent coppice rotations result in increased biomass yields).

Table 4.3. Emission factors for biomass harvesting and processing activities for coppice irrigated treatment [kg CO_2/ha/yr]

	Treatment Ci				
Rotation	1st	2nd	3rd	4th	5th
Harvester	78.23	79.80	80.58	81.36	82.14
Trailer+blower	19.56	19.95	20.14	20.34	20.54

4.4.3.2 Biomass transport emissions

The emissions from transportation of biomass were calculated using diesel fuel consumption rates and emission factors for diesel fuel. In the calculation of the emission factors for diesel fuel we included the emissions due to the three key greenhouse gases CO_2 (3.663 kg CO_2/l diesel), CH_4 (0.00012 kg CH_4/l diesel) and N_2O (0.0000082 kg N_2O/l diesel) (SGA Energy Ltd. 2000, Environment Canada , (S&T)2 Consultants Inc. 2010). To bring the CH_4 and N_2O emission factors to CO_2 equivalence, we used global warming potential (GWP) factors of 25 for CH_4 and 298 for N_2O for a 100-year horizon (IPCC 2007).

Using the same assumptions as in section 1.4.2 about transportation distance, length of segments for dirt/gravel/paved roads, and hourly diesel fuel consumption rates, we calculated a diesel fuel consumption rate by log trucks of 0.13 l/dry tonne/km and an emission factor for logs transportation of 0.48 kg CO_2/dry tonne/km. For chip trucks the calculated diesel fuel consumption rate was 0.15 l/dry tonne/km and the emission factor for chips transportation was 0.55 kg CO_2/dry tonne/km.

4.4.3.3 Emissions from biorefinery operations

Estimates of GHG emissions that occur during the biochemical conversion processes at the biorefinery vary substantially in the literature. We examined studies that would report the actual CO_2 emissions during the biochemical conversion process without subtracting any credit for the carbon contained in the biomass processed – as we account separately for that carbon in the Bio-CarbD model.

The GHGenius model ((S&T)2 Consultants Inc. 2011) reports the emissions factor for ethanol production from white chips to be 1.3 kg CO_2/l EtOH, using the energetic emission factor of 54.4 kg CO_2/GJ EtOH and the volumetric ethanol energy density of 0.023579 GJ/l EtOH on higher heating value basis ((S&T)2 Consultants Inc. 2011). This includes a credit of 0.3 kg CO_2/l EtOH (12.6 kg CO_2/GJ EtOH) for using the lignin to generate electricity that is sold on the grid and is presumed to displace fossil-fuel based electricity. The credit represents 19% of the production emissions.

California EPA (2009) reports the emissions factor for ethanol production to be 2.3 kg CO_2/l EtOH (99.3 kg CO_2/GJ EtOH). This includes a credit of 0.2 kg CO_2/l EtOH (10.2 kg CO_2/GJ EtOH) for co-generation of surplus electricity from lignin, which represents 9% of the production emissions.

National Research Council (2009) reports a range for the emissions factor for ethanol production from white chips: 3.0 – 4.3 kg CO_2/l EtOH (129.1 – 182.1 kg CO_2/GJ EtOH), based on major (high) to little (low) improvements in technology and process efficiency.

For the test case we assumed an emissions factor for ethanol production from white chips of 1.4 kg CO_2/l EtOH (58.3 kg CO_2/GJ EtOH; for low emissions scenarios) and 4.3 kg CO_2/l EtOH (183.7 kg CO_2/GJ EtOH; for high emissions scenarios). This included a credit for co-generation of surplus electricity from lignin: 0.4 kg CO_2/l EtOH (low emissions scenarios) and 0.06 kg CO_2/l EtOH (high emissions scenarios). We assumed that the exported energy would replace electricity that would have been produced by the average provincial mix (*i.e.* we used the GHGenius values for British Columbia, Canada).

4.4.3.4 Embodied emissions from construction of machinery, as a proportion of machinery use from machinery lifetime

Different types of machinery and vehicles are used for mechanized forest operations and transportation of biomass; production of these machinery and vehicles leads to CO_2 emissions. However, not all these CO_2 emissions can be attributed to the biofuel project, since the lifetime (hours or kilometres) of machinery and vehicles is expected to be greater than the quantity (hours or kilometres) used or spent for the biofuel project.

This approach is consistent with recently developed and approved methodologies such as the Roundtable for Sustainable Biofuels (RSB) which include emissions from infrastructure in their proposed GHG calculation methodology (Roundtable on Sustainable Biofuels 2011, Roundtable on Sustainable Biofuels 2011). Infrastructure includes farm equipment (e.g., tractors), fossil feedstock production equipment (e.g., drilling equipment), fuel production equipment (e.g., refineries), and other. The standards and certification system of the RSB has been recognized by the European Commission (European Commission 2011) as a way to demonstrate and document compliance with the EU biofuels mandate.

For the test case we included in the greenhouse balance calculations the emissions embodied in the construction of machinery and vehicles, more specifically the proportion of hours/kilometres that the machinery was used in the biofuel project, from the machinery total lifetime hours/kilometres.

The embodied emissions were calculated differently for: a) machinery used on a per hectare basis (*i.e.* tractors, boom sprayer), b) machinery used on a per wood quantity basis (*i.e.* feller, skidder, loader, grinder, delimber), and c) trucks used on a per kilometre basis, as follows.

a) $EMB\_ha = EMB\_unit * MCH\_weight * MCH\_prodha^{-1} * MCH\_life^{-1}$ [kg CO_2/ha].
b) $EMB\_kg = EMB\_unit * MCH\_weight * MCH\_prodkg^{-1} * WOOD\_proc \ \ MCH\_life^{-1}$ [kg CO_2/ha].
c) $EMB\_km = EMB\_unit * MCH\_weight * MCH\_capkm^{-1} * WOOD\_proc$ $DIST\_proj * DIST\_life^{-1}$ [kg CO_2/ha].

where,

EMB_unit = unit embodied emission factor for machinery and trucks; constant [kg CO_2/kg].

MCH_weight = machine weight; constant [kg].

MCH_prodha = machine productivity on a per hectare basis; constant [ha/hr].

MCH_life = machine lifetime hours; constant [hr].

MCH_prodkg = machine productivity on a per wood quantity basis; constant [kg wood/hr].

WOOD_proc = unit quantity of wood processed by respective machine; variable by treatment [kg wood/ha].

MCH_capkm = unit truck capacity; constant [kg wood].

DIST_proj = distance for transport of biomass, return trip from biorefinery to forest site harvested; variable by location of land unit [km].

DIST_life = expected truck lifetime distance; constant [km].

Table 4.4 shows the technical data assumptions for calculation of embodied emissions in machinery and trucks.

Table 4.4. Technical data assumptions for calculation of embodied emissions in machinery and trucks

		EMB_unit [kg CO_2/kg]	MCH_weight [kg]	MCH_prodha [ha/hr]	MCH_prodkg [kg wood/ha]	MCH_capkm [kg wood]	MCH_life [hr]	DIST_life [km]
Per ha basis use	Tractor 160[1]	1.34[2]	6,620[3]	2.50[4]	--	--	8,000[5]	--
	Tractor 130[2]	1.34	4,820[3]	4.17[4]	--	--	8,000[5]	--
	Boom sprayer[3]	1.34	7,718[6]	13.39[4]	--	--	8,000[5]	--
Per kg wood basis use	Feller	1.34	36,590[7]	--	15,750[7]	--	18,000[8]	--
	Skidder	1.34	15,200[7]	--	15,750[7]	--	10,000[8]	--
	Loader	1.34	26,900[7]	--	15,750[7]	--	15,000[8]	--
	Grinder	1.34	31,320[7]	--	15,848[7]	--	19,360[7]	--
	Delimber	1.34	3,175[9]	--	15,848[8]	--	10,000[8]	--
	Harvester	1.34	11,380[10]	--	44,657[11]	--	15,000[8]	--
	Blower-tractor-trailer	1.34	6,620[8]	--	44,657[11]	--	10,000[8]	--
Per km basis use	Truck	1.34	15,900[7]	--	--	18,750[8]	--	1,000,000[7]

[1] With V-Ripper 25” O.C., 10ft attachment (ripping)

[2] Calculated assuming (a) machine main components: 0.45 kg steel/kg machine, 0.45 kg cast iron/kg machine, and 0.10 kg rubber/kg machine (Gautam *et al.* 2010) and (b) emission factors for the production of component materials: 1.46 kg CO_2/kg steel, 1.35 kg CO_2/kg iron (IPCC 2006), and 0.74 kg CO_2/kg rubber (Sullivan *et al.* 2010).

[3] RitchieSpecs http://www.ritchiespecs.com/

[4] (Lazarus 2011)

[5] (Edwards 2009, Scown et al. 2012, FAO 1992, Caterpillar Inc. 2011)

[6] (Deere & Company)

[7] (Gautam *et al.* 2010)

[8] Assumed in this study

[9] (Danzco Inc.)

[10] (New Holland North America, Inc. 2002)

[11] (Buchholz and Volk 2011)

4.4.3.5 Embodied emissions from construction and maintenance of biorefinery facilities

Similar to the construction of machinery and trucks described in the section above, a biorefinery will also need to be built as part of the biofuel project. The production, transport and installation of all the biorefinery component machinery and infrastructure lead to CO_2 emissions. These emissions are assumed to not have occurred in the absence of the biofuel project, as the biorefinery was built for the specific purpose of the project.

The EcoInvent database reports an emissions factor of 4,571,500 kg CO_2 for "ethanol hydrolysis and fermentation plant" infrastructure, for an ethanol plant with an annual capacity of 90,000 tons ethanol over a lifetime of 20 years (Frischknecht *et al.* 2007, Roundtable on Sustainable Biofuels 2011). The embodied emissions factor from construction of biorefinery facilities for the test case was calculated to be 9,833,131 kg CO_2 for an annual production capacity of 227.12 million litres EtOH/year using a scaling factor of 0.7 to account for economies of scale (*i.e.* assuming that higher capacity facilities have lower unit emissions on a per litre of EtOH basis).

During the economic life of the biorefinery plant, the installations and facilities need to be repaired, maintained or replaced as needed. These operations lead to CO_2 emissions. We follow Gautam *et al.* (2010) and assume that operations and maintenance work is performed every year to the extent that at the end of the economic life all facilities and machines have been updated and/or renewed. The yearly emissions rate associated with operations and maintenance was calculated by dividing the infrastructure construction emissions by the number of years of the facility economic life; *i.e.* for a 30-year lifetime, the rate is 3.33% per year. For the test case we calculated an emissions factor for maintenance of the biorefinery plant of 0.0433 kg CO_2/l EtOH.

It is assumed that similar facility-related emissions from gasoline production system operations (including gasoline refinery operations) are also included in the respective emissions factor for gasoline.

4.4.3.6 Emissions from ethanol transport, storage, distribution and dispensing

The emission factor for transport, storage, distribution and dispensing of ethanol is reported in the literature as low as 0.03 kg CO_2/l EtOH (1.4 kg CO_2/GJ EtOH) in GHGenius ((S&T)2 Consultants Inc. 2010), and as high as 0.06 kg CO_2/l EtOH (2.7 kg CO_2/GJ EtOH) in California EPA (2009), representing 2.1% and 2.5% respectively of the production emissions from conversion to ethanol activities in those reports.

For the test case we assumed an emissions factor for transport, storage, distribution and dispensing of ethanol of 0.06 kg CO_2/l EtOH (2.5 kg CO_2/GJ EtOH), representing 2.1% of the conversion production emissions.

4.4.3.7 Emissions from final use of ethanol

In this study it is assumed that the final use of ethanol is in internal combustion engines of automobiles. On average, CO_2 emissions represent 95-99% of the total greenhouse gas emissions from a passenger vehicle (U.S. EPA 2011). Since CO_2 represents almost all GHG emissions from final use of ethanol, and since all the emissions (*i.e.* carbon loss) from the removal of harvested biomass (which in part is converted to ethanol) are already included in the Bio-CarbD model, the emissions from the final use of ethanol were not added directly to the lifecycle GHG balance in order to avoid double counting.

The CO_2 emissions from final use of ethanol are indeed included in the GHG balance calculations, represented by the loss of carbon in the harvested biomass that is converted to ethanol.

4.4.3.8 Additional notes

The diesel CH_4 and N_2O emissions were added to the diesel CO_2 emissions. Where needed, the carbon quantities were converted to CO_2 equivalence (CO_2-eq) by multiplying carbon estimates by 3.67, which is the ratio of the molecular weight of CO_2 (44) to the molecular weight of Carbon (12). CO_2-eq was reported in metric tonnes per hectare or in total tonnes. All emissions results are reported in CO_2-eq.

4.4.4 Planning horizon

The planning horizon considered in our analysis was 100 years. In CHAPTER 3 the choice of this planning horizon length was discussed in more detail. In year 1 the land units selected for land-use change are prepared, and planting starts the following year. A period of 8 years is considered to allow the planted biomass to reach harvesting age (which in the test case was a maximum of 8 years), and to build the new biorefinery. The next nine decades represent three economic lifetimes for the biorefinery, which is reported or assumed in the literature to be around 30 years (Aden *et al.* 2002). For the test case it was assumed that after 30 years of service either another biorefinery will either be constructed or, more likely, the existing one would be updated/maintained to expand its life for another 30 years.

4.4.5 The baseline (business as usual) case

To compare the GHG balance of the biofuel project with the displaced emissions from a reference fossil fuel system, we used the principle of additionality. We considered all the GHG emissions and carbon sequestration that were additional to the business-as-usual case, *i.e.* those that would not have occurred in the absence of the biofuel project. Only the additional emissions from construction and use of machinery, trucks and biorefinery (or, more precisely, the proportion of emissions attributed directly to the existence of the biofuel project) were included.

From a biogenic carbon perspective it was assumed that, in the absence of the biofuels project, the baseline biogenic carbon balance of all land units would be constant, in both biomass and soil. In other words, the carbon balance of the land units would have been in equilibrium in the absence of the biofuel project, *i.e.* the carbon absorbed from the atmosphere is equivalent with the carbon emitted back into the atmosphere through natural decay processes. Even if this assumption may not reasonably represent the actual condition of the landbase, this was done mainly to provide a baseline for comparison. However, for practical applications where the biogenic carbon dynamics in the absence of the project is known, the C3BO model can easily accommodate the comparison with an actual baseline.

4.4.5.1 Gasoline production, distribution, and combustion emissions

The emission factor for gasoline included emissions from (1) the combustion at final use (*i.e.* in internal combustion engines), and (2) the production and distribution activities of the gasoline system. The combustion emission factor for gasoline was assumed to be 2.30 kg CO_2/litre gasoline (Environment Canada 2010, (S&T)2 Consultants Inc. 2010), as the sum of emission factors for the three key greenhouse gases: 2289 g CO_2/l, 0.14 g CH_4/l, and 0.022 g N_2O/l. The emission factors for CH_4 and N_2O were brought to CO_2 equivalency by multiplying the above values with their respective GWP values on a 100-year basis. The production and distribution emission factor for gasoline is reported as 0.98 kg CO_2/l gasoline (27.912 kg CO_2/GJ, HHV basis) in GHGenius for Western Canada ((S&T)2 Consultants Inc. 2010).

For the test case we assumed a total gasoline emission factor of 2.52 kg CO_2/l gasoline (low emissions scenarios) and 3.30 kg CO_2/l gasoline (high emissions scenarios), which compares to the range of values reported in the literature: from 2.46 kg CO_2/l gasoline by Sheehan *et al.* (2004) cited in Adler (2007), to 3.16 kg CO_2/l gasoline (U.S. DOE-NETL 2008) and 3.26 kg CO_2/l gasoline in Farrell *et al.* (2006) and O'Hare *et al.* (2009).

4.4.5.2 Energy efficiency of ethanol compared with gasoline

When comparing the emissions from ethanol use with those of gasoline, it is important to take into account the difference in energy efficiency for the two fuels. If an ethanol volume is assumed to displace an equivalent amount of gasoline for a particular end use, the two fuels can be compared based on their respective unit energy content, or on consumption of fuel (litres per 100 km, or km per litre) (Gnansounou *et al.* 2009).

On a high heating value basis, the volumetric energy density of ethanol is 23.6 MJ/l EtOH, while that of gasoline is 34.7 MJ/l gasoline ((S&T)2 Consultants Inc. 2010). Similarly, on a low heating value basis, the volumetric energy density of ethanol is 21.2 MJ/l EtOH, while that of gasoline is 31.9 MJ/l gasoline (Gnansounou *et al.* 2009). The energy efficiency factor of ethanol compared with gasoline is therefore assumed to be 0.68 l gasoline/l EtOH, or 1.50 l EtOH/l gasoline. This means that, from an energy content perspective, roughly 50% more volume of ethanol is needed to displace the same amount of energy from a volume of gasoline.

However, ethanol is expected to displace gasoline in transportation vehicles only as a blend. Typically, gasoline blends containing up to 10% ethanol (by volume) are used in Canada (Environment Canada 2010). However, flex fuel vehicles can use a blend of up to 85% ethanol and 15% gasoline (E85).

Since the E10 blend contains less energy per unit of volume than gasoline alone (E0), it is expected that it will have lower fuel efficiency than gasoline. Knoll *et al.* (2009) report an average reduction of 3.7% in fuel efficiency of E10 compared with E0. The energy efficiency of E0 compared with E10 is therefore assumed in the test case to be 0.96 litres E0/litres E10.

4.4.6 Experimental scenarios for C3BO model

To demonstrate the Carbon Balance and Biomass to Biofuel Optimization planning system, C3BO, we used the general settings for the test case as described above, and then selected four experimental scenarios with different model inputs and configurations.

The four experimental scenarios were generated using two sets of variables: (1) unit cost of ethanol production activities, including biomass production and conversion to ethanol; and (2) GHG emissions from mechanized production activities, and carbon removal and releases between the project and the atmosphere. The resulting scenario matrix is shown in Table 4.5.

Table 4.5. Scenario matrix

	Low unit cost of ethanol production	High unit cost of ethanol production
Low GHG emissions	Scenario 1	Scenario 2
High GHG emissions	Scenario 3	Scenario 4

The unit ethanol production cost and the GHG emissions were chosen to define the four scenarios because the C-BOS model optimizes only for cost but not for GHG emissions, and therefore the scenario results could potentially indicate whether the C-BOS model reacts in some way due to increased costs that would lower the GHG emissions. For example, the four scenarios selected could answer the question: is there an indication of correlation between high costs and low GHG emissions? It is noted that a more comprehensive sensitivity analysis will be reported in CHAPTER 5.

For all scenarios two land unit types were available, one non-irrigated (LU1) and one irrigated (LU2). For the non-irrigated land unit, LU1, two treatments were available, a single stem treatment (S) with rotation of 8 years and a coppice treatment (C) with 3 rotations of 5 years each (for a total 15 year rotation). For the irrigated land unit, LU2, two treatments were available, a single stem irrigated treatment (Si) with rotation of 6 years and a coppice irrigated treatment (Ci) with 5 rotations of 3 years each (for a total 15 year rotation).

Each land unit type had an unlimited available planting area, and the project planning horizon was 100 years. Non-production treatments were available at the beginning of the project. To simplify the analysis, in the experimental scenarios no idle treatments were allowed at the end of the planning horizon.

Key model input variables were set in each scenario at either the lowest or the highest value in their respective ranges (Table 4.6). The choice for an input variable to be set at either a low or high value was determined by its anticipated effect on cost or emissions. For example, lower project costs are anticipated for high biomass MAI, for low transportation distance, and for high conversion efficiency.

Table 4.6. Selection of low and high levels for input variables, by scenario

		Scenario 1	Scenario 2	Scenario 3	Scenario 4
Biomass MAI	[t wood/ha/yr]	High	Low	High	Low
Biomass transportation distance	[km]	Low	High	Low	High
Conversion efficiency	[l EtOH/t wood]	High	Low	High	Low
Biorefinery capacity	[l EtOH]	High	Low	High	Low
Enzyme unit cost	[$/kg enzyme]	Low	High	Low	High
Electricity price	[$/kWh]	High	Low	High	Low
Initial carbon stocks	[t C/ha]	Low	Low	High	High
Loss of DOM carbon in harvest years	[%]	Low	Low	High	High
DOM carbon loss at land-use change	[%]	Low	Low	High	High
EtOH production emissions	[kg C/l EtOH]	Low	Low	High	High
Gasoline production emissions	[kg C/ l gasoline]	High	High	Low	Low
Blend of ethanol in gasoline		Low	Low	High	High
Irrigation emissions	[kg C/ha]	Low	Low	High	High

Table 4.7. Values of input variables for test case scenarios

		Scenario 1	Scenario 2	Scenario 3	Scenario 4
Biomass MAI, treatment S	[t wood/ha/yr]	15.7	11.0	15.7	11.0
Biomass MAI, treatment Si	[t wood/ha/yr]	20.2	14.1	20.2	14.1
Biomass MAI, treatment C	[t wood/ha/yr]	15.7	11.0	15.7	11.0
Biomass MAI, treatment Ci	[t wood/ha/yr]	20.2	14.1	20.2	14.1
Biomass transportation distance	[km]	40	200	40	200
Conversion efficiency, white chips	[l EtOH/t wood]	299	234	299	234
Conversion efficiency, whole tree chips	[l EtOH/t wood]	268	209	268	209
Biorefinery capacity	[l EtOH]	378,541,178	75,708,236	378,541,178	75,708,236
Enzyme unit cost	[$/kg enzyme]	2.48	8.00	2.48	8.00
Electricity price	[$/kWh]	0.115	0.050	0.115	0.050
Initial carbon stocks, vegetation above-ground	[t C/ha]	7.0	7.0	41.5	41.5
Initial carbon stocks, DOM	[t C/ha]	23.8	23.8	141.1	141.1
Loss of DOM carbon in harvest years: snag, medium, and slow-decaying pools	[%]	0%	0%	10%	10%
Loss of DOM carbon in harvest years: fast-decaying pools	[%]	0%	0%	15%	15%
Loss of DOM carbon in harvest years: very fast-decaying pools	[%]	0%	0%	20%	20%
DOM carbon loss at	[%]	0%	0%	25%	25%

		Scenario 1	Scenario 2	Scenario 3	Scenario 4
land-use change					
EtOH production emissions	[kg CO_2/l EtOH]	1.4	1.4	4.3	4.3
Gasoline production emissions	[kg CO_2/ l gasoline]	3.3	3.3	2.5	2.5
Blend of ethanol in gasoline		E10	E10	E85	E85
Irrigation emissions	[kg CO_2/ha]	37	37	342	342

* Biomass yield is for stem, bark and branch

For comparison with the findings from previous chapters, the eight scenarios of CHAPTER 2 have been also analyzed from the net GHG balance perspective.

4.5 Results and discussion

The biofuel project GHG balance results for scenarios 1-4 are shown in Figure 4.3 through Figure 4.6. To help with interpretation, the orange areas represent GHG emissions and the green areas represent GHG credits and/or carbon sequestration. All quantities in Figure 4.3 through Figure 4.6 have been described earlier in equations 4.1-4.9. For example, in Figure 4.3 $CSEQ_{cr}$ has a value of -55 kg CO_2/GJ EtOH and is calculated as per equation 4.7 as the sum of $CATM_{cr}$ (-560 kg CO_2/GJ EtOH), GHG_pCNV_{em} (290 kg CO_2/GJ EtOH), and GHG_DCAY_{em} (214 kg CO_2/GJ EtOH). Similarly, the $netGHG\_PROJ_{em}$ value of -67 kg CO_2/GJ EtOH is calculated as the sum of GHG_PROJ_{em} (28 kg CO_2/GJ EtOH) and $FOSS_{cr}$ (-95 kg CO_2/GJ EtOH). The values that are zero in Figure 4.3 through Figure 4.6 are in fact non-zero; the disparity is due to rounding.

As expected, the “low GHG emission” scenarios (*i.e.* the scenarios that had input parameters in such a way as to potentially result in low net GHG balance) resulted in much fewer emissions than the “high GHG emission” scenarios. The life cycle net GHG emissions of the ethanol production system in scenario 1 are -67 kg CO_2/GJ EtOH, which means that the biofuel project emissions are lower than those of the displaced gasoline with 67 kg CO_2/GJ EtOH. Similarly, for scenario 2, the net GHG emissions of the biofuel project are -52 kg CO_2/GJ EtOH (*i.e.* ethanol has 52 kg CO_2/GJ EtOH less emissions than gasoline). In contrast, scenarios 3 and 4 result in higher ethanol emissions than those of displaced gasoline. In scenario 3, the ethanol project has 156 CO_2/GJ EtOH more emissions than the baseline fossil fuel system, and in scenario 4 the ethanol emissions are 227 CO_2/GJ EtOH.

The “low cost” scenarios (*i.e.* scenarios 1 and 3 with input parameters in such a way as to potentially lead to low ethanol production unit costs) resulted in both low and high GHG balances. Similarly, the “high cost” scenarios 2 and 4 lead to both low and high GHG balances. The results indicate that the optimization model does not seem to react in some way to increased costs that would lower the net GHG balance.

The high biomass yield and conversion efficiency (scenarios 1 and 3) resulted in both low and high emissions, as did the low biomass yield and conversion efficiency (scenarios 2 and 4). Similarly, the high transportation distance (scenarios 2 and 4) lead to both high and low GHG emissions, and the low transportation distance had the same result.

The results did not indicate any obvious correlation between C3BO model inputs (*e.g.*, biomass yield, conversion efficiency, transportation distance, biorefinery capacity) and the net GHG balance. This provides an impetus for a more detailed sensitivity analysis, which will be conducted in CHAPTER 5.

SCENARIO 1

Net GHG emissions (netGHG_PROJem Year 100)	-67
Gross GHG emissions (GHG_PROJem)	28
Production emissions (GHG_PRODem)	83
Init. biomass removal emiss. (GHG_iLUCem)	3
Init. mech. activities emiss. (GHG_iMECem)	0
Proj. mech. activities emiss. (GHG_pMECem)	7
Init. transport emiss. (GHG_iTRNem)	0
Proj. transport emiss. (GHG_pTRNem)	16
Biorefinery infrastr. constr. emiss. (GHG_iINFRem)	0
Init. biorefinery activities emiss. (GHG_iREFem)	0
Init. conversion emiss. (GHG_iCNVem)	0
Init. lignin use credit (iLIGcr)	0
Proj. biorefinery activities emiss. (GHG_pREFem)	58
Proj. conversion emiss. (GHG_pCNVem)	87
Proj. lignin use credit (pLIGcr)	-29
Carbon sequestration credit (CSEQcr)	-55
C from atmosphere credit (CATMcr)	-560
Proj. biomass removal emiss. (GHG_pBMHem)	290
DOM decay emiss. (GHG_DCAYem)	214
Credit substitution fossil fuel (FOSScr)	-95
Init. biofuel produced credit (iFOScr)	-1
Proj. biofuel produced credit (pFOScr)	-94

Figure 4.3. GHG balance of the biofuel project (kg CO_2/GJ EtOH), 100-year horizon, scenario 1

SCENARIO 2

Net GHG emissions (netGHG_PROJem Year 100)	-52
Gross GHG emissions (GHG_PROJem)	44
Production emissions (GHG_PRODem)	118
Init. biomass removal emiss. (GHG_iLUCem)	5
Init. mech. activities emiss. (GHG_iMECem)	0
Proj. mech. activities emiss. (GHG_pMECem)	10
Init. transport emiss. (GHG_iTRNem)	0
Proj. transport emiss. (GHG_pTRNem)	45
Biorefinery infrastr. constr. emiss. (GHG_iINFRem)	0
Init. biorefinery activities emiss. (GHG_iREFem)	0
Init. conversion emiss. (GHG_iCNVem)	0
Init. lignin use credit (iLIGcr)	0
Proj. biorefinery activities emiss. (GHG_pREFem)	58
Proj. conversion emiss. (GHG_pCNVem)	86
Proj. lignin use credit (pLIGcr)	-29
Carbon sequestration credit (CSEQcr)	-74
C from atmosphere credit (CATMcr)	-752
Proj. biomass removal emiss. (GHG_pBMHem)	370
DOM decay emiss. (GHG_DCAYem)	308
Credit substitution fossil fuel (FOSScr)	-95
Init. biofuel produced credit (iFOScr)	-1
Proj. biofuel produced credit (pFOScr)	-94

Figure 4.4. GHG balance of the biofuel project (kg CO_2/GJ EtOH), 100-year horizon, scenario 2

SCENARIO 3

Net GHG emissions (netGHG_PROJem Year 100)	156
Gross GHG emissions (GHG_PROJem)	229
Production emissions (GHG_PRODem)	218
Init. biomass removal emiss. (GHG_iLUCem)	16
Init. mech. activities emiss. (GHG_iMECem)	0
Proj. mech. activities emiss. (GHG_pMECem)	6
Init. transport emiss. (GHG_iTRNem)	0
Proj. transport emiss. (GHG_pTRNem)	15
Biorefinery infrastr. constr. emiss. (GHG_iINFRem)	0
Init. biorefinery activities emiss. (GHG_iREFem)	4
Init. conversion emiss. (GHG_iCNVem)	5
Init. lignin use credit (iLIGcr)	-2
Proj. biorefinery activities emiss. (GHG_pREFem)	176
Proj. conversion emiss. (GHG_pCNVem)	264
Proj. lignin use credit (pLIGcr)	-87
Carbon sequestration credit (CSEQcr)	11
C from atmosphere credit (CATMcr)	-541
Proj. biomass removal emiss. (GHG_pBMHem)	280
DOM decay emiss. (GHG_DCAYem)	272
Credit substitution fossil fuel (FOSScr)	-73
Init. biofuel produced credit (iFOScr)	-3
Proj. biofuel produced credit (pFOScr)	-70

Figure 4.5. GHG balance of the biofuel project (kg CO_2/GJ EtOH), 100-year horizon, scenario 3

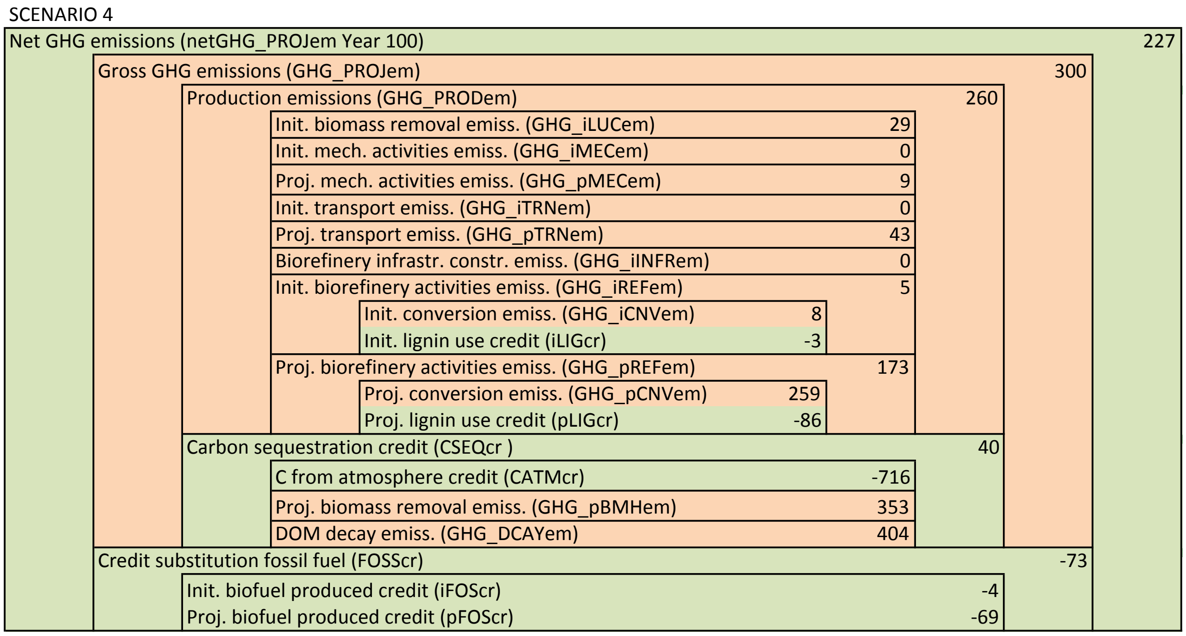

Figure 4.6. GHG balance of the biofuel project (kg CO_2/GJ EtOH), 100-year horizon, scenario 4

The number of hectares needed for the biofuel project, as well as other biomass and ethanol production unit costs are shown in Table 4.8.

Table 4.8. C-BOS results for experimental scenarios

	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Objective value [mil. $]	14,625	5,348	14,625	5,348
Total ethanol produced [l x10^6]	23,091	4,618	23,091	4,618
Plantation Area Needed [ha]	96,356	35,222	96,356	35,222
Biomass Harvest-Process Cost Total [$/t]	69.24	82.28	66.82	78.24
Land Rent Cost Total [$/t]	13.90	19.86	13.90	19.86
Biomass Transportation Cost Total [$/t]	29.37	57.20	29.37	57.20
Total Biomass Delivered Cost [$/t]	112.52	159.34	110.10	155.30
Total Biomass Delivered Cost [$/litre]	0.38	0.68	0.37	0.67
Refinery Cost [$/litre]	0.29	0.72	0.29	0.72
Total EtOH Production Cost [$/litre]	0.66	1.40	0.65	1.38

The net GHG emissions of the biofuel project are shown in Table 4.9, and the GHG savings (calculated using equation 4.10) of the biofuel project compared with the displaced fossil fuel system are summarized in Table 4.10.

Table 4.9. Net GHG emissions of biofuel project

[kg CO_2/GJ EtOH] or [g CO_2/MJ EtOH]	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Gross GHG emissions (GHG_PROJem)	28	44	229	300
Credit substitution fossil fuel (FOSScr)	-95	-95	-73	-73
Net GHG emissions (netGHG_PROJem Year 100)	-67	-52	156	227

The biofuel project as considered in scenario 1 and 2 would result in 70% and 54% GHG savings respectively, compared with the displaced fossil fuel system (*i.e.* the baseline). In other words, the existence of a biofuel project as described in scenario 1 would result in 70% less GHG emissions than what would have happened in its absence. A project with conditions as assumed in scenario 2 would result in 54% less GHG emissions than its absence. This suggests that biofuel projects that are set up with characteristics as detailed in scenarios 1 and 2 would represent viable climate change mitigation strategies (from the perspective of GHG emissions and carbon sequestration).

In contrast, biofuel projects that are set up as per conditions in scenarios 3 and 4 would produce much more GHG emissions than what would have happened in their absence (*i.e.* more than the baseline). The biofuel project in scenario 3 would result in 215% more GHG emissions, and in scenario 4 would result in 312% more.

It is interesting to note that whether the GHG savings are high or low does not seem to be associated to either low or high ethanol production costs. In the low cost scenarios 1 and 3, the GHG savings can be both high (scenario 1) and low (scenario 3). Similarly, in the high cost scenarios 2 and 4, the GHG savings can be both high (scenario 2) and low (scenario 4).

Table 4.10. GHG savings and total ethanol production cost of the biofuel project

	GHG_Savings	Total EtOH Production Cost [$/litre]
Scenario 1	70.4%	0.66
Scenario 2	54.2%	1.40
Scenario 3	-215.3%	0.65
Scenario 4	-312.4%	1.38

Figure 4.7 through Figure 4.10 show the GHG emissions and credits of the biofuel project in each scenario. The largest credit ($CATM_{cr}$) is given for the carbon absorbed from the atmosphere through biomass growth. In scenarios 1 and 2 the largest source of GHG emissions is from the biomass harvested (GHG_pBMH_{em}), most of which is converted to ethanol and the remainder is lost (*i.e.* oxidized) during processing. In scenarios 3 and 4, two other emissions sources are also important: biorefinery activities (GHG_pCNV_{em}), and natural decay of organic matter (GHG_DCAY_{em}).

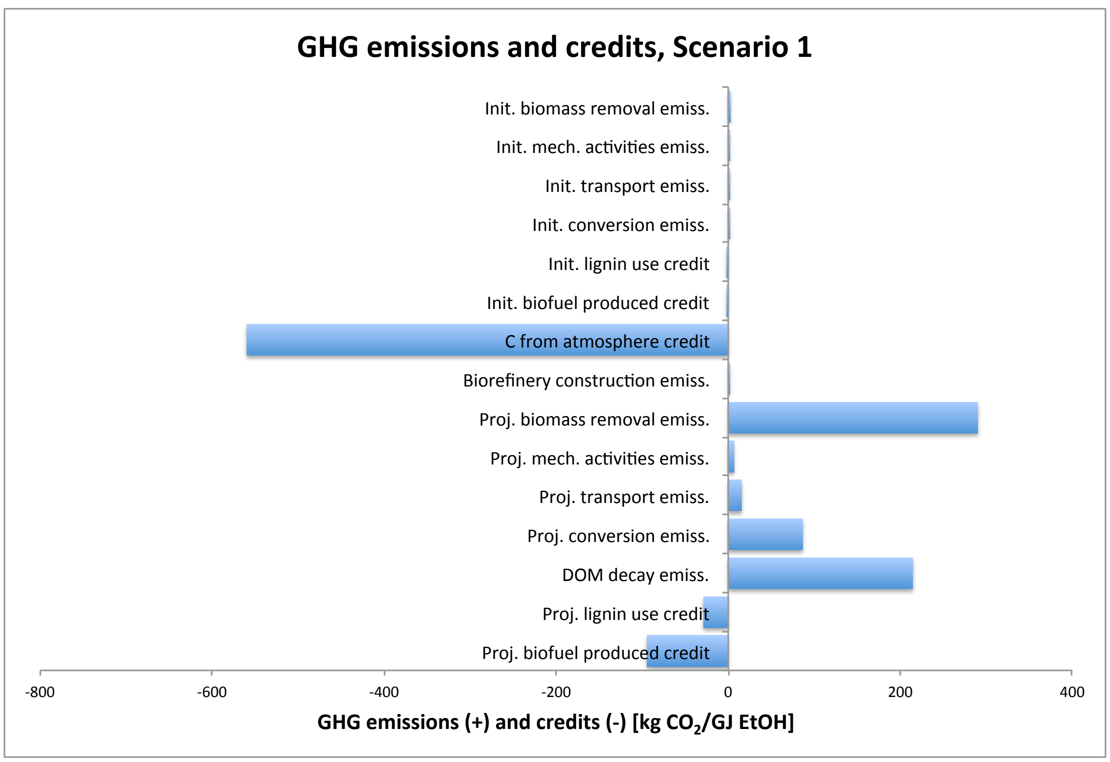

Figure 4.7. Project GHG emissions and credits, 100-year horizon, scenario 1

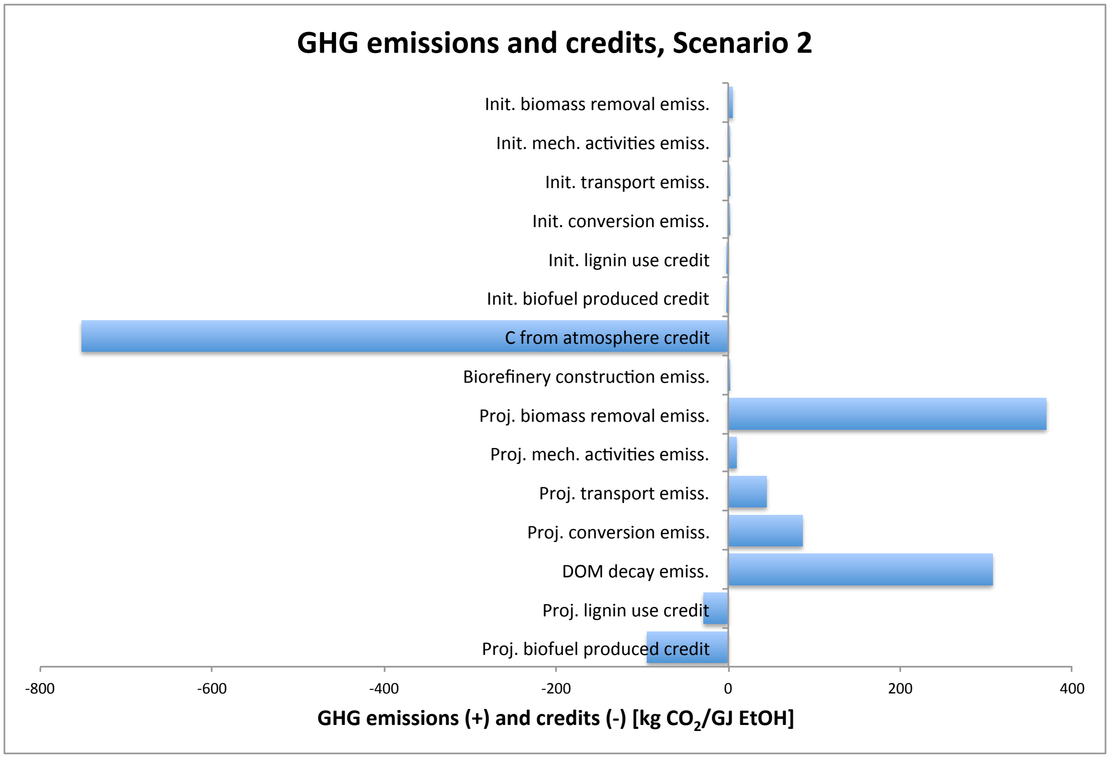

Figure 4.8. Project GHG emissions and credits, 100-year horizon, scenario 2

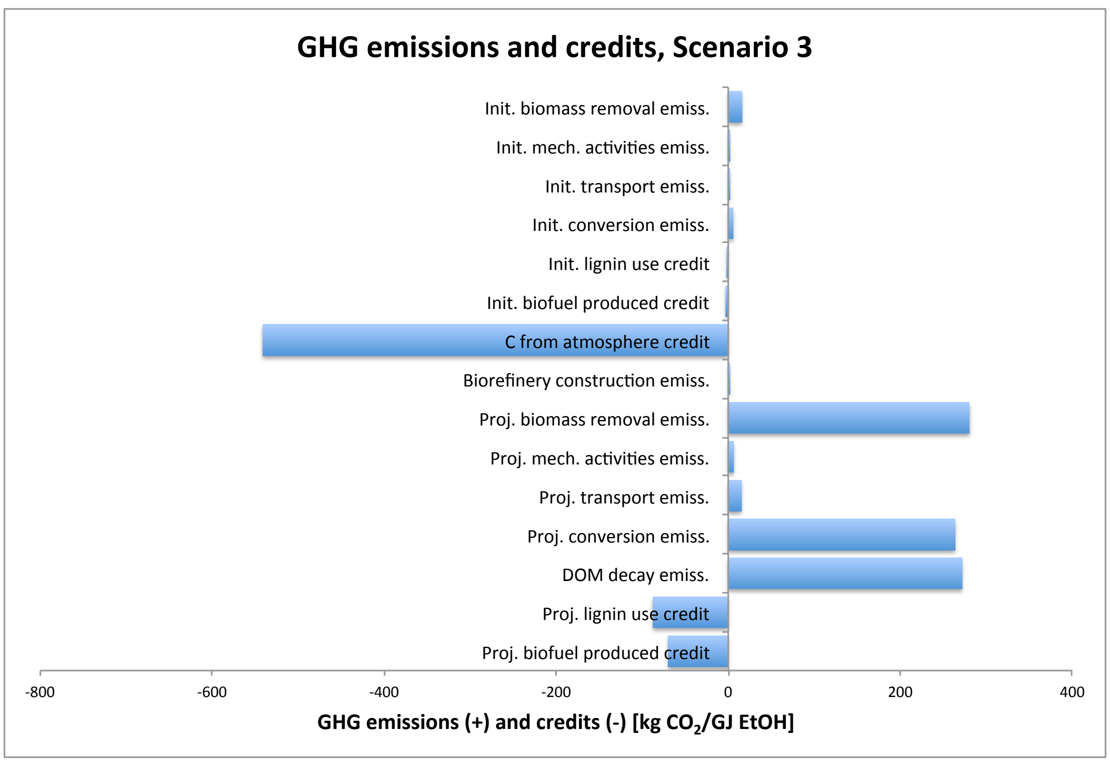

Figure 4.9. Project GHG emissions and credits, 100-year horizon, scenario 3

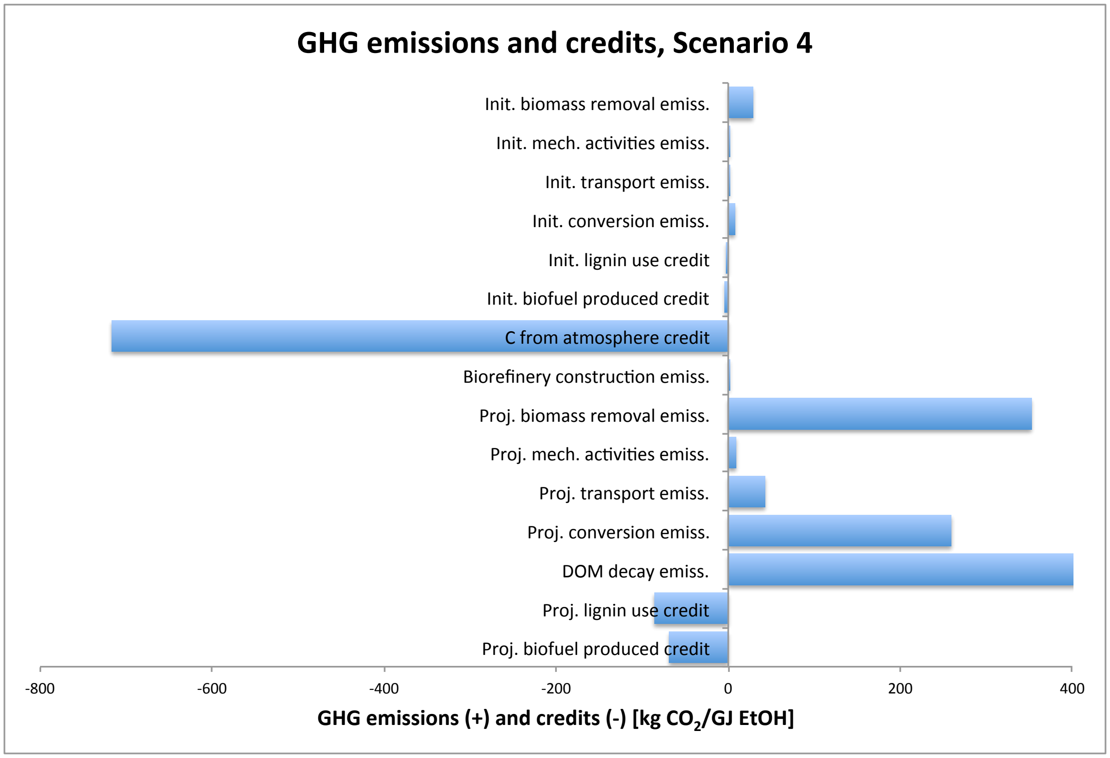

Figure 4.10. Project GHG emissions and credits, 100-year horizon, scenario 4

The dynamics of GHG emissions of the biofuel project are shown in Figure 4.11 for all four scenarios. The Y axis of the graph represents the life cycle net GHG emissions balance ($netGHG\_PROJ_{em}$). In scenarios 1 and 2 the GHG emissions start in year 0 (at land-use change) with some emissions from initial land-use change (*i.e.* less than 5 kg CO_2/GJ EtOH) and after roughly year 4 the project GHG balance becomes increasingly negative, meaning that the biofuel project produces fewer emissions than the displaced gasoline system. Scenarios 1 and 2 have fairly similar GHG balance dynamics throughout the years, with the difference between them growing larger towards the end of the project horizon. The fact that there are two crossover points between scenarios 1 and 2 is due to the different shapes that the GHG balance dynamics curve can take, depending on the project specific conditions; an example of two very different GHG balance curves are shown in Figure G.1 and Figure G.2, where it can be seen that the rates of change in the GHG balance can be quite different early in the project as carbon accumulates in live pools and DOM, as well as later in the project as the GHG emissions can be less or more than the GHG credits.

In scenarios 3 and 4 the emissions at land-use change (year zero) are higher, then until year 8 biomass is growing so carbon is removed from atmosphere and sequestered in live biomass; after year 8 the project GHG emissions are increasingly larger than those of the displaced gasoline, and by year 100 the emissions of the biofuel project are substantially larger than the baseline.

Since the four scenarios of the test case have been intentionally selected to represent extreme low GHG emissions or high emissions of the biofuel project, it is conceivable that any other combinations of model input parameters would result in net GHG emissions somewhere between 70.4 kg CO_2/GJ EtOH and -312.4 kg CO_2/GJ EtOH (see APPENDIX G for two additional examples of possible scenarios). There is no economic incentive in the optimization model for lower GHG emissions, and therefore it is not unexpected that the low cost scenarios 1 and 3 resulted in both low and high net GHG balances. Similarly, the high cost scenarios 2 and 4 also led to both low and high net GHG balances.

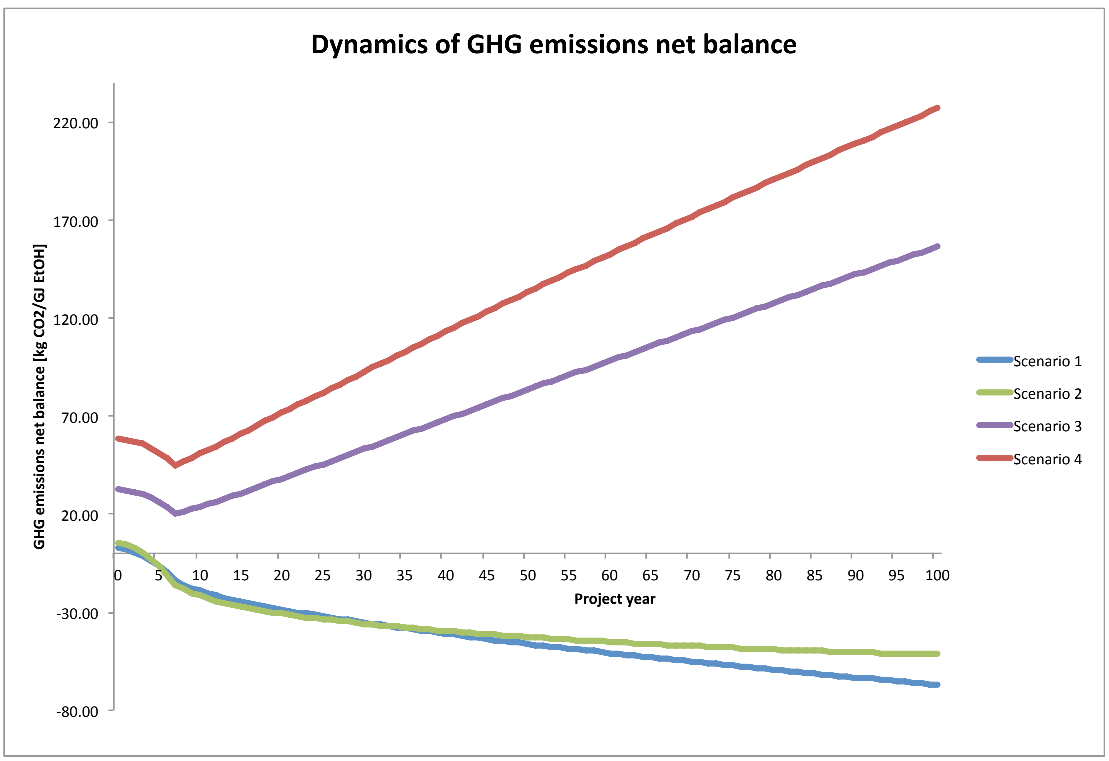

Figure 4.11. Net GHG emissions balance dynamics (netGHG_PROJ$_{em}$), scenarios 1-4, 100-year horizon

The C3BO model accounts for all the changes in biogenic carbon pools through time, ensuring that the carbon balance (the carbon absorbed from atmosphere must equal the carbon sequestered in biomass and DOM, plus the carbon released back into atmosphere) is maintained at project level and also on an annual basis.

Table 4.11 through Table 4.14 show the biogenic carbon balances for all scenarios, expressed as content of carbon dioxide per unit of ethanol energy [kg CO_2/GJ EtOH]. The results can be interpreted as follows, for scenario 1 for example (Table 4.11): prior to land-use change (*i.e.* before the project), the existing land units contained 4 kg CO_2/GJ EtOH sequestered in live biomass and 10 kg CO_2/GJ EtOH in DOM. At the moment of land-use change (year -1) the existing live biomass is removed as part of site preparation activities: 3 kg are removed as harvested biomass, and 1 kg stays on site and is transferred to the DOM pools as dead foliage, branches and roots. During the following 100 years of the project, carbon is continually captured by growing biomass (for a total of 560 kg CO_2/GJ EtOH), and is then partly released back when that biomass is harvested and partly transferred to DOM during growth and at harvest times.

In scenario 1, at the end of the final year of the project the balances of sequestered carbon are: 15 kg CO_2/GJ EtOH sequestered in live biomass in various stages of growth, but which will not have been harvested by the end of the project; and, 51 kg CO_2/GJ EtOH that have accumulated in DOM pools. The life cycle project balance of sequestered carbon is 52 kg CO_2/GJ EtOH (11 = 15 – 4 in live biomass; and 41 = 51 – 10 in DOM).

The life cycle project balance for the carbon released is 507 kg CO_2/GJ EtOH, of which 293 kg CO_2/GJ EtOH are from harvested biomass (released back into atmosphere either as ethanol oxidation emissions, or biomass losses/oxidation from manufacturing and production processes, and 214 kg CO_2/GJ EtOH released back to atmosphere through natural decay processes of organic matter.

After accounting for all the biogenic carbon captured from, or released to, the atmosphere in scenario 1, there is a net positive carbon balance (*i.e.* sequestered and stored on the land units) of 52 kg CO_2/GJ EtOH. In scenario 2 the balance is even "better" at 69 kg CO_2/GJ EtOH. For scenarios 3 and 4 (the "high GHG emissions" scenarios) the biofuel project actually results in a net loss of biogenic carbon stored on the land units (-27 and -69 kg CO_2/GJ EtOH respectively) compared with the absence of the project.

Table 4.11. Carbon balance in biogenic carbon pools, scenario 1 [kg CO_2/GJ EtOH]

SCENARIO 1		Prior to LUC	Year -1 (LUC)	Year 100	Life cycle project balance
C_SEQUESTERED	C_SEQ_LBIO	4	0	15	52
	C_SEQ_DOM	10	11	51	
C_RELEASED	C_HBIO	0	3	293	507
	C_DCAY	0	0	214	
C_ABSORBED	C_ABSORBED	0	0	-560	-560
TOTAL		14	14	14	0

Table 4.12. Carbon balance in biogenic carbon pools, scenario 2 [kg CO_2/GJ EtOH]

SCENARIO 1		Prior to LUC	Year -1 (LUC)	Year 100	Life cycle project balance
C_SEQUESTERED	C_SEQ_LBIO	7	0	20	69
	C_SEQ_DOM	18	20	74	
C_RELEASED	C_HBIO	0	5	375	682
	C_DCAY	0	1	308	
C_ABSORBED	C_ABSORBED	0	0	-752	-752
TOTAL		25	25	25	0

Table 4.13. Carbon balance in biogenic carbon pools, scenario 3 [kg CO_2/GJ EtOH]

SCENARIO 1		Prior to LUC	Year -1 (LUC)	Year 100	Life cycle project balance
C_SEQUESTERED	C_SEQ_LBIO	22	0	15	-27
	C_SEQ_DOM	58	48	38	
C_RELEASED	C_HBIO	0	16	296	568
	C_DCAY	0	16	272	
C_ABSORBED	C_ABSORBED	0	0	-541	-541
TOTAL		79	79	79	0

Table 4.14. Carbon balance in biogenic carbon pools, scenario 4 [kg CO_2/GJ EtOH]

SCENARIO 1		Prior to LUC	Year -1 (LUC)	Year 100	Life cycle project balance
C_SEQUESTERED	C_SEQ_LBIO	39	0	19	-69
	C_SEQ_DOM	104	86	55	
C_RELEASED	C_HBIO	0	29	381	785
	C_DCAY	0	28	404	
C_ABSORBED	C_ABSORBED	0	0	-716	-716
TOTAL		143	143	143	0

The importance of DOM decay emissions to the net GHG balance of the biofuel project is illustrated in Figure 4.12, which shows the GHG emissions and GHG credits/carbon sequestration for each scenario. The value of the DOM decay emissions is (much) larger in absolute terms than the net GHG balance for all scenarios (e.g. 214 *vs*. 67 kg CO_2/GJ EtOH in scenario 1, and 404 *vs*. 227 kg CO_2/GJ EtOH in scenario 4).

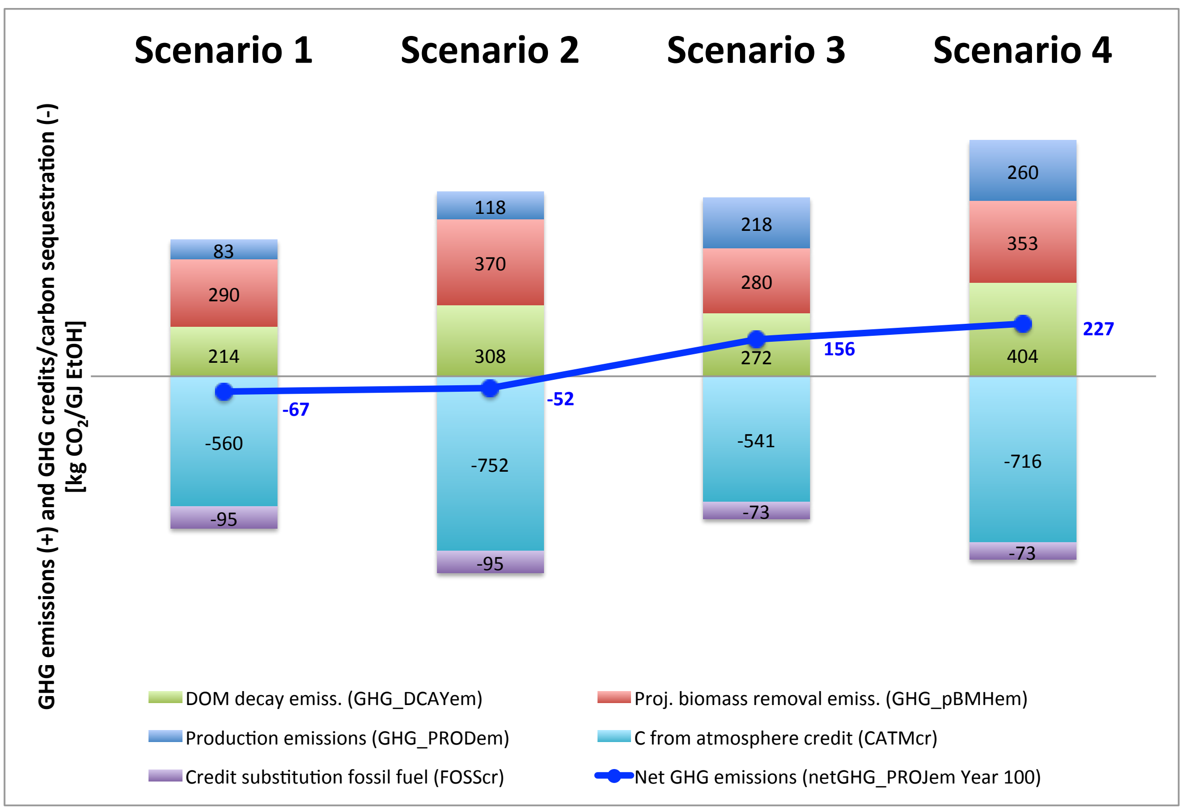

Figure 4.12. GHG emissions and GHG credits/carbon sequestration, showing magnitude of DOM decay emissions

Figure 4.12 shows that in scenarios 1 and 2 the sum of biomass removal emissions (*i.e.* harvested biomass) and DOM decay emissions is lower than the carbon removed from the atmosphere by the project. This helps explain why the net GHG balance of scenarios 1 and 2 are negative (*i.e.* the ethanol production system has lower emissions than the displaced gasoline production system). In contrast, in scenarios 3 and 4 the sum of biomass removal emissions (*i.e.* harvested biomass) and DOM decay emissions is higher than the carbon removed from the atmosphere by the project. This is explained by the large DOM decay emissions, due to i) the large value of initial DOM pools (*i.e.* 104 kg CO_2/GJ EtOH prior to land-use change in scenario 4), ii) a portion of DOM carbon is assumed to be released at land-use change (from 104 to 86 kg CO_2/GJ EtOH in scenario 4), and iii) the emission factors are applied to the remaining large DOM pool.

This is further illustrated in Figure 4.13 that shows the biogenic carbon releases (positive values) and removals (negative values) from atmosphere. The blue squares represent the net biogenic carbon balance for each scenario. In scenarios 1 and 2 the net biogenic carbon balance is negative, meaning that the terrestrial carbon pools (above- and below-ground) have increased as a result of the project. However, in scenarios 3 and 4 there is less biogenic carbon stored in pools at the end of the project compared with the beginning of the project. Again, this is due to the large carbon removals from DOM pools through natural decomposition processes.

The importance of including the emissions from DOM decay in the overall biogenic carbon balance is evident in scenarios 3 and 4, where the emissions from biomass removal and from DOM decay exceed the capture of carbon dioxide from atmosphere. This means that the project not only does not sequester any net amount of carbon in live or dead organic matter pools at the end of the time horizon, but actually releases a fraction of the carbon that was stored in the DOM pools before land-use change to the atmosphere.

If the biomass had been simply considered “carbon-neutral”, then none of the emissions from DOM decay would have been considered. On one hand, the analysis would not have identified that the project actually results in net emissions in scenarios 3 and 4, from the biogenic carbon perspective.

On the other hand, an assumption of biomass "carbon neutrality" would not have identified that in scenarios 1 and 2 the project actually results in a net sequestration of carbon in live and dead organic matter pools.

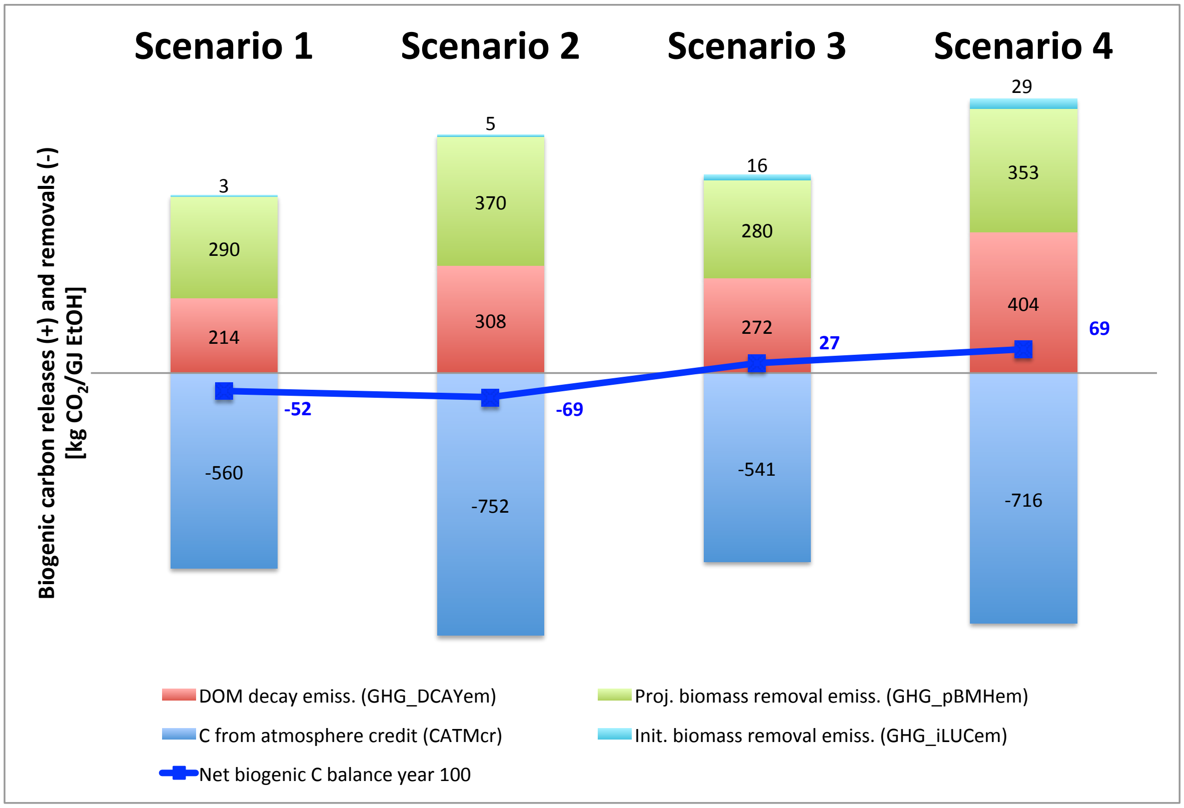

Figure 4.13. Biogenic carbon releases (positive values) and removals (negative values) from atmosphere

4.5.1 Analysis of scenarios from chapter 2

For comparison with the findings from previous chapters, the eight scenarios of CHAPTER 2 have been also analyzed from the net GHG balance perspective.

Similar to the findings of CHAPTER 3, the largest net GHG balance was in Scenario 2B, where the project "sequestered" 8,222,872 more tonnes of carbon (equivalent to 92 kg CO2/GJ EtOH) compared with the status-quo (the displaced fossil fuel system). In Scenario 2B, the ethanol production cost is higher by $1,508,138,589 and the sequestered carbon is higher by 8,222,872 tonnes. This is equivalent to a differential cost of $887/tonne of carbon, which makes scenario 2B unlikely to be pursued as a carbon sequestration strategy.

It is reasonable to expect that biofuel project proponents will pursue instead the lowest cost option in scenario 4A.

Table 4.15. C3BO results for the eight scenarios of CHAPTER 2

		Scenario 1A	Scenario 2A	Scenario 3A	Scenario 4A	Scenario 1B	Scenario 2B	Scenario 3B	Scenario 4B
C-BOS objective function value	[$ x10^6]	8,222	7,869	7,327	7,033	8,441	8,460	7,905	8,220
Ethanol unit production cost	[$/l]	0.90	0.87	0.83	0.81	0.91	0.90	0.86	0.84
Total ethanol produced	[l x10^6]	13,855	13,855	13,855	13,855	13,855	14,122	14,083	15,411
Average annual area	[ha/yr]	79,873	71,079	70,997	63,947	84,456	75,606	75,593	72,635
Net biogenic C sequestered	[t C x10^6]	7.05	7.33	6.48	6.73	8.70	8.74	7.33	8.27
Net GHG balance	[t C x10^6]	6.67	7.05	6.45	6.81	8.29	8.58	7.45	7.74

4.6 Limitations of the study

Since the C3BO model incorporates the C-BOS and Bio-CarbD models, it is reasonable to expect that the limitations of these two models can influence the overall limitations of the C3BO model.

From a computational standpoint, a disadvantage of the Type I model formulation used in the C-BOS model, is the potentially large size of the LP problem. The number of model variables can be very large because the number of possible regeneration-harvesting sequences for all the land parcels increases as the planning horizon expands, *i.e.* the decision tree shown in APPENDIX B becomes very large very quickly as more years and the respective possible regeneration-harvesting sequences are added to the planning horizon. However, since this model is not intended to be used for real-time decisions, and the model solve time is only several minutes, this limitation is not expected to significantly impact the model usage.

Another limitation of the C-BOS model formulation, inherent to linear programming techniques, is that it is deterministic. That is, all the possible treatments and their associated growth and yield tables, production costs, must be specified *a priori*. For example, uncertainty is not considered specifically in the model formulation; it is assumed that the expectation of the growth and yield assumptions adequately capture the inherent risk of reduced yields due to various unfavourable conditions.

The linearity of the objective function can also be considered a limitation of the C-BOS model. For example, it is assumed that the improvements in biomass yield and in conversion efficiency result in proportional increases or decreases in the cost of various production activities that are dependent on the respective yields.

One of the limitations of the Bio-CarbD model is the reference of CBM-CFS3 parameters (*i.e.* carbon transfer matrix coefficients, transfer and decay parameters) specified for hardwood species, which are assumed to be suitable for this study. This aspect could be improved in the future as research with more appropriate parameters are published.

An important component of planning biomass production over a long time horizon is uncertainty. Life-cycle greenhouse-gas-performance estimates of second-generation biofuels remain uncertain in the absence of large-scale crop production trials and commercial-scale biorefineries (IEA 2010). The Bio-CarbD model assumes that the biomass growth and yield and the conversion efficiency are within the assumed ranges. Similarly, for the biorefinery operations and ethanol production emissions, they are assumed to be within certain ranges referenced from the available literature. As new data becomes available the assumptions modelled in this study can be easily updated.

An important area for further research is to investigate the effect of uncertainty and variation in input parameters on wood-to-ethanol production planning, which is explored using sensitivity analysis in CHAPTER 5.

4.7 Conclusion

From a carbon and GHG balance perspective, the viability of displacing fossil-derived transportation fuels with biofuels from wood biomass is largely debated. This study proposes a new method for analyzing the viability of biofuel projects, the Carbon Balance and Biomass to Biofuel Optimization planning model (C3BO).

The primary goal of C3BO is to determine the life cycle net greenhouse gas emissions directly generated by a biofuel project (from the initial land-use change to biomass production, construction of biorefinery, ethanol production, to final consumption in internal combustion engine, for a set planning horizon) and compare these GHG emissions with what might have happened in the absence of the project (*i.e.* with the baseline of using an equivalent quantity of fossil-derived fuels, and not using the lands). A secondary goal is to provide insight on the production costs of ethanol (including production costs of biomass and transport), to calculate the necessary land area, and to determine the optimal sequence of planting/harvesting activities that minimizes biomass production costs.

A novel approach of the C3BO model framework is linking the planting/harvesting optimization model CBOS (that minimizes biomass production costs activities) with the biogenic carbon balance model Bio-CarbD (that uses the activities sequence output from CBOS for each plantation land parcel, and determines the biogenic carbon dynamics for biomass and soil in each land parcel). The combination of the two models in one framework permits forecasting of the impacts of different biomass and ethanol production strategies not only on the biogenic carbon dynamics, but also on the overall life-cycle project net GHG balance.

Other novel outcomes of the C3BO model are:

- a project-level methodology that facilitates inquiry about the GHG impacts of a biofuel project compared with the absence of the project,
- a consideration of initial land-use change effects not only on the live biomass carbon pools, but also on DOM and soil carbon pools; modeling the dynamics of soil biogenic carbon sequestration and emissions for each land parcel area over time,
- modeling the potential affect of harvesting/planting mechanized activities on soil carbon emissions through soil compaction and scarification,
- modeling the sequestration of biogenic carbon not only in live biomass, but also in DOM pools; this can reveal situations when the biofuel project results in net losses (emissions) of biogenic carbon, but also in net gains (sequestration) in biogenic carbon, results that might be overlooked by simply assuming that biomass is "carbon neutral,"
- inclusion of both i) carbon removals from atmosphere through biomass growth and ii) biogenic carbon releases back into atmosphere through biomass removal/harvesting, eliminates the need to assume the disputed notion of whether biomass is carbon neutral or not.

To illustrate the utility of the C3BO model we provide results for a test case with four experimental scenarios using a prototype wood-to-ethanol project in order to verify and validate the modeling framework. The low- and high-emissions, and low- and high-costs scenarios analyzed in this study produced a wide range of results. The life cycle net GHG balance of wood-to-ethanol projects ranged from being largely negative (meaning that the ethanol project emissions are much lower than those of the displaced gasoline reference system), to being largely positive (the ethanol project emissions are much larger than what have happened in the absence of the project), or somewhere in between. It is important to note that the conditions (input parameters) of the four scenarios were not chosen to represent typical or average conditions; the conditions were chosen deliberately towards the possible extremes so that the potential range of emissions is observed.

A comparison with other studies is shown in Figure 4.14. While it is difficult to make a direct comparison, since the system boundaries and assumptions may not be similar, it can be seen that the C3BO test case results include the range of results from other studies, except for the very low values reported for cellulosic ethanol by Farrell *et al.* (2006).

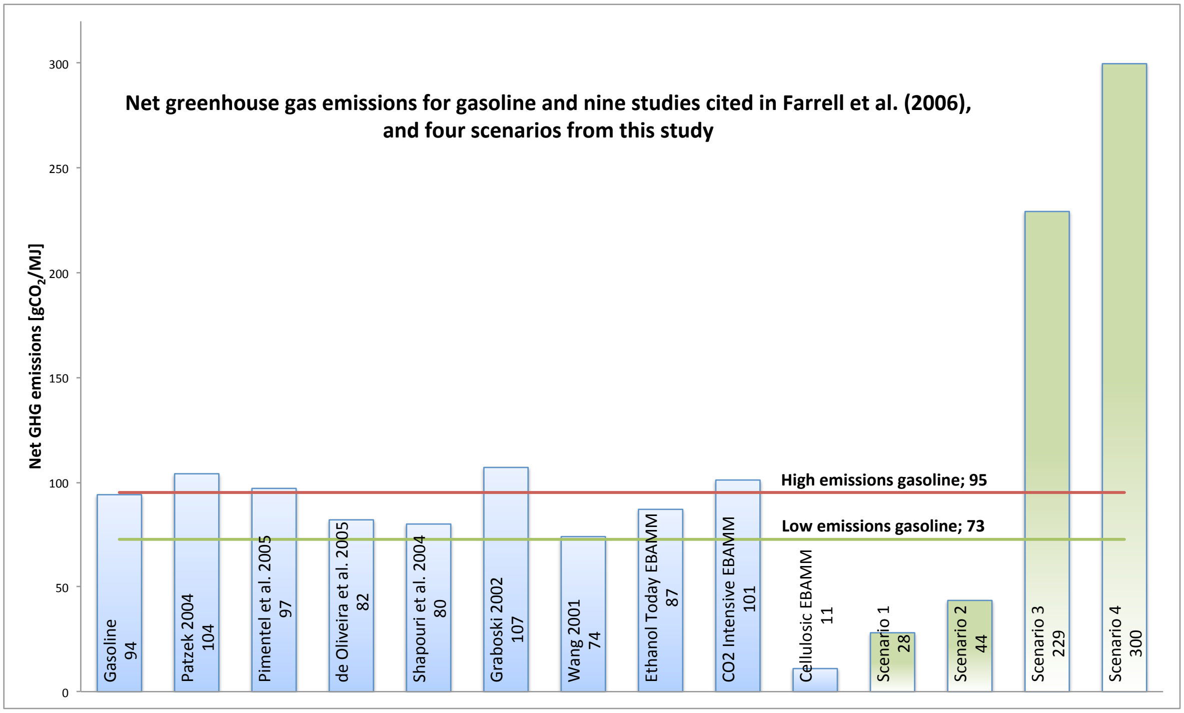

Figure 4.14. Net GHG emissions of biofuel project for i) gasoline and nine studies cited in Farrell *et al.* (2006), and ii) scenarios 1 through 4 from this chapter (100-year horizon).

The two horizontal lines in Figure 4.14 represent the lower and upper bounds of gasoline emissions considered in this study (green for low, 73 gCO2/MJ; red for high, 95 gCO2/MJ). The model results show that the potential direct land-use change impacts of biofuel projects (including the initial biomass removal but perhaps more importantly the affect of the project on the organic matter balance and emissions in the soil) can be an important component of the life-cycle GHG balance, and therefore it indicates that they need to be included in GHG balance analyses of biofuel projects. Model results suggest that it takes several decades for a biofuel project to offset the initial emissions from land-use change, through carbon sequestration in biomass and soil, and displacement of fossil fuel emissions. Results also indicate that the magnitude of the reductions in emissions, compared with the fossil fuel baseline, is highly dependent on the length of the time considered for the biofuel project; *e.g.* project emissions at 60 years can be substantially different than at 100 years.

Previous reports on life cycle GHG emissions of ethanol projects found either that displacing gasoline with ethanol would result in less GHG emissions, or that it would result in more GHG emissions, than the absence of the ethanol project. On one side of the argument are the studies who suggest that wood ethanol projects have a lower GHG emissions balance than the displaced gasoline, and the question is not "if" ethanol is better than gasoline but "by how much". In contrast, there are the studies that argue that ethanol may result in higher emissions than the displaced fossil fuel.

The findings of this study suggest that both positions are correct: biofuel projects can indeed produce fewer emissions than the fossil fuel baseline under specific conditions, but they can also result in more emissions under other specific conditions.

The conditions under which ethanol projects that displace gasoline can be viable climate mitigation strategies will be analyzed in CHAPTER 5.

CHAPTER 5 Wood-to-ethanol bioenergy from fast growing tree plantations: a sensitivity analysis of project net greenhouse gas balance, ethanol production costs, and plantation area

5.1 Synopsis

From the perspective of mitigating the increase of atmospheric greenhouse gases biofuel projects can be considered viable strategies, if they have net a GHG balance that is lower than that of the displaced fossil fuel under business as usual. In the previous chapters our analysis has shown that displacing gasoline with wood ethanol from fast growing tree plantations may not always be a viable climate mitigation strategy. Due to the uncertainty of many factors that influence the GHG balance of biofuels projects, an important question is to what extent the outcomes are influenced by the input data.

To determine the impact of input variables on the net GHG balance, an extensive sensitivity analysis of the outputs from the Carbon Balance and Biomass to Biofuel Optimization planning model (C3BO) was carried out, and forms this chapter. Since the land available for establishment of dedicated plantations is limited, and the production cost of ethanol is an important factor in determining the feasibility of displacing gasoline, the sensitivity analysis is also carried out for these two key outputs from the C3BO model.

The sensitivity analysis was carried out using two different approaches: 1) changing one-factor-at-a-time (OFAT) and quantify what effect this produces on each of the outputs, and 2) fitting a multiple linear regression to the model responses and using relative importance metrics as direct measures of sensitivity.

When a biofuel project has a very low net GHG balance, changes in ethanol conversion efficiency and in gasoline production emissions have the most impact on the GHG balance, followed by ethanol production emissions. *Ceteris paribus*, a 10% improvement in conversion efficiency results in a 4.1% – 6.2% increase in net GHG balance. However, when a biofuel project has a very high GHG balance, the impact of changes in biomass yield is large, similar to that of ethanol conversion efficiency. A 10% improvement in biomass yield will result in a reduction in net GHG balance of 4.3% – 6.7%, depending on the project time horizon. And, a 10% improvement in conversion efficiency results in an increase of net GHG between 4.8% and 6.0%.

The sensitivity analysis using linear regression suggests that the net GHG balance is most sensitive to ethanol production emissions and initial carbon stocks, and much less sensitive to the other input variables. As the project time horizon increases, the impact of initial carbon stocks decreases, while ethanol production emissions increases.

For ethanol production cost, the linear regression sensitivity analysis suggests that the variability in ethanol cost is most sensitive to the variability in conversion efficiency. A 10% improvement in ethanol conversion efficiency results in a 5.3-5.7% reduction in ethanol production cost, while a 10% improvement in biomass yield reduces ethanol cost by 2.1-2.2%.

Regarding the plantation area required, the linear regression sensitivity analysis for the range of input values suggests that the variability in plantation area is most sensitive to the biorefinery capacity, and less sensitive to biomass yield and conversion efficiency. However, one-factor-at-a-time changes in all the three input variables produce similar changes in plantation area. A 10% improvement in either biomass yield or conversion efficiency results in a 9.1% decrease in plantation area, and a 10% increase in biorefinery capacity results in a 10% increase in area.

5.2 Introduction

In order to mitigate the increase of atmospheric carbon dioxide, displacing transportation fuels such as gasoline with biofuels from sustainably grown wood biomass is one of the climate strategies considered by policy makers. The actual emissions associated with ethanol production from woody biomass sources are uncertain as these processes are still under development (McKechnie *et al.* 2011). Currently, there are no fully operational commercial large-scale wood ethanol biorefineries, and no large scale dedicated plantations of fast growing trees that supply directly and exclusively such large biorefineries.

In the absence of real production systems to study, the viability of wood ethanol from fast growing plantations to displace gasoline is typically evaluated by either extrapolating experimental results from laboratory or small-scale operations, or by systems modeling using theoretical simulation models. This dissertation has employed both these approaches in the development of the Carbon Balance and Biomass to Biofuel Optimization planning model (C3BO) to determine the net GHG balance of biofuel systems.

Since the specific parameters of the real large-scale biomass- and biofuel-production systems cannot be measured directly, there is considerable uncertainty around the actual values that the various input parameters can take, and there is a high probability that the true GHG emissions for a specific system will be substantially different from the 'default results' (Cherubini *et al.* 2009). It therefore becomes necessary to study how the results are influenced by changes in the assumptions regarding the input data, and how the model outputs can be related to the inputs. In other words, which factors of production are likely to have the largest impact on GHG emissions, cost and other important performance measures? This information may help drive future research and investment.

The impact of model inputs on the outputs of biofuel production models has been reported by many studies in the peer-reviewed literature. Schlamadinger and Marland (1996) report that the biomass growth rate is of utmost importance to the 100-years carbon balance for short-rotation forestry projects, and that fossil fuel inputs (from biomass production, harvesting, transport, biofuel conversion) have a relatively small impact. Marland and Schlamadinger (1997) analyzed biofuels from managed forests and found that the net C sequestration potential is very sensitive to the efficiency with which forest products substitute for alternate fuels, and to the forest growth rate. McKechnie *et al.* (2011) studied biofuels from managed forests and report that forest carbon dynamics are significant to biofuel greenhouse gas balance. Farrell *et al.* (2006) analyzed the net energy of ethanol and concluded that they are most sensitive to assumptions about co-product allocation, and that agricultural practices (e.g., conservation tillage) significantly impact GHG emissions. Searchinger *et al.* (2008) and Cherubini *et al.* (2009) report that emissions from land-use change have a significant impact on greenhouse gas balance of biofuels. Gnansounou *et al.* (2009) suggest that the life cycle assessment of greenhouse gas balance of biofuels is highly sensitive to the method used to allocate the impacts between the co-products, the type of reference systems, the choice of the functional unit, and the type of blend. Singh *et al.* (2010) suggest that life cycle analysis results of lignocellulosic ethanol are more sensitive to the changes in parameters related to the biomass and ethanol yield.

This dissertation chapter contributes to the discussion, by comparing the impact of input factors for different project-specific situations (*i.e.* low and high project emissions, ethanol cost and area) and for different project horizons, thus aiding to identify what source of uncertainty weighs more on the current study's conclusions and to understand the viability of displacing fossil-derived fuels with wood ethanol from dedicated fast growing tree plantations.

5.3 Methodology

As mentioned earlier, the sensitivity analysis of the input variables on model outputs was carried out using two different approaches: 1) changing one-factor-at-a-time (OFAT) and quantify what effect this produces on each of the outputs, and 2) fitting a multiple linear regression to the model responses and using relative importance metrics as direct measures of sensitivity. The assumptions and settings used in the C3BO model are similar to those used in the analysis of CHAPTER 4, except that only one treatment was considered for biomass growth strategies: single stem non-irrigated (S). The choice of treatment type was arbitrary, however, of the most common production methods of wood for transportation biofuels cited in the literature.

5.3.1 One-factor-at-a-time

The impact of input factors on GHG balance, ethanol cost, and area was analyzed by changing one factor at a time while keeping all others constant, for six scenarios (Table 5.1):

- Scenario 1: low project net GHG balance
- Scenario 2: high project net GHG balance
- Scenario 3: low ethanol production cost
- Scenario 4: high ethanol production cost
- Scenario 5: low plantation area needed
- Scenario 6: high plantation area needed

The results from CHAPTER 3 and CHAPTER 4 suggest that the project time horizon has an influence on carbon and GHG balances, and therefore the sensitivity analysis for project net GHG balance was done separately for project durations of 38, 68, and 100 years, for both scenarios 1 and 2.

Table 5.1. Reference values of input variables for test case scenarios

		Sc. 1	Sc. 2	Sc. 3	Sc. 4	Sc. 5	Sc. 6
Biomass MAI [t wood/ha/yr]		15.7	11.0	15.7	11.0	15.7	11.0
Biomass transportation distance [km]		40	200	40	200	40	40
Conversion efficiency, white chips [l EtOH/t wood]		299	234	299	234	299	234
Biorefinery capacity [l EtOH *1000]		227	227	379	76	379	76
Initial carbon stocks, vegetation above-ground [t C/ha]		7.0	41.5	7.0	7.0	7.0	7.0
Initial carbon stocks, DOM [t C/ha]		23.1	136.7	23.1	23.1	23.1	23.1
Loss of DOM carbon in harvest years: [%]	snag, medium, and slow-decaying pools	0%	10%	0%	0%	0%	0%
	fast-decaying pools	0%	15%	0%	0%	0%	0%
	very fast-decaying pools	0%	20%	0%	0%	0%	0%
DOM carbon loss at land-use change [%]		0%	25%	0%	0%	0%	0%
EtOH production emissions [kg CO_2/l EtOH]		1.4	4.3	1.4	1.4	1.4	1.4
Gasoline production emissions [kg CO_2/ l gasoline]		3.3	2.5	3.3	3.3	3.3	3.3
Blend of ethanol in gasoline		E15	E85	E85	E10	E10	E10

The 3 variables related to loss of DOM carbon in harvest years, due to scarification of soil caused by mechanized activities, have been combined together for this analysis, meaning that they were all increasing or decreasing by the same proportion (*i.e.* either all were at minimum, all were at maximum, or all were at the same relative point between min and max within their respective ranges). For this reason, the loss of DOM carbon in harvest years was considered to be only one input variable.

In each scenario, C3BO simulations were repeated for selected model input parameters with the parameter systematically shifted plus and minus 10% and 50% from their reference values, *ceteris paribus*. For some input parameters decreases were not feasible (e.g., factors that had zero as reference value). For high ethanol conversion efficiency the 50% increase was not feasible (*i.e.* beyond the theoretical maximum), so in this case plus and minus 20% shifts were analyzed instead.

5.3.2 Multiple linear regression analysis

To produce the data for the multiple linear regression analysis, a number of 45 replications was performed so that the degrees of freedom were at least 30 (Crawley 2005), *i.e.* 45-11-1 = 33 ≥ 30 d.f. Random values were generated for each of the 11 input variables within their respective ranges mentioned in Table 5.1.

To answer the question of which independent variables have the most affect on the variability of each of the three dependent variables (ethanol production cost, plantation area, and net GHG balance), stepwise regression analyses were conducted to choose the models that were able to predict each dependent variable, from a set of candidate models. The 11 input (independent) variables were chosen for the initial model based on their potential influence on the dependent variable, as follows:

- For cost of ethanol: v01 biomass yield (mean annual increment; t wood/ha/year); v02 biomass transportation distance (km); v03 conversion efficiency (l EtOH/t wood); v04 biorefinery production capacity (l EtOH/year); v05 blend of ethanol in gasoline (l gasoline /l EtOH);
- For plantation area: v01 biomass yield (mean annual increment; t wood/ha/year); v02 biomass transportation distance (km); v03 conversion efficiency (l EtOH/t wood); v04 biorefinery production capacity (l EtOH/year); v05 blend of ethanol in gasoline (l gasoline /l EtOH);
- For net GHG balance: v01 biomass yield (mean annual increment; t wood/ha/year); v02 biomass transportation distance (km); v03 conversion efficiency (l EtOH/t wood); v04 biorefinery production capacity (l EtOH/year); v05 blend of ethanol in gasoline (l gasoline /l EtOH); v06 initial carbon stocks in biomass and soil (t C/ha); v07 loss of soil carbon (slow decaying pools) from mechanized activities in harvest years (%); v08 ethanol production emissions (kg C/l EtOH); v09 gasoline

production emissions (kg C/l gasoline); v10 soil carbon los at land-use change (%); v11 irrigation emissions (kg C/ha).

The statistical model for predicting each of the three dependent variables was chosen in the stepwise regression analysis using the Akaike Information Criterion (AIC) (Akaike 1974) procedure in the statistical software R (R Core Team 2012). The stepwise regression procedure kept only the statistically significant variables in the final model. Multiple linear regression analyses for the chosen models were conducted to determine if each of the dependent variables could be predicted from the selected independent variables. Regression equations were derived for each of the dependent variables.

Analyses of variance were conducted in order to show whether there were significant effects or not. The assumptions of linearity, homoscedasticity, and normality were verified using graphical plots and statistical tests. The data were screened for completeness and violation of assumptions prior to analysis.

Finally, the quantification of individual regressors' contribution (also known as relative importance) to the multiple regression models was explored with the LMG[7] (Lindeman *et al.* 1980) and PMVD[8] (Feldman 2005) metrics, as recommended by Grömping (2006). The analysis was performed using the "relaimpo" package in the statistical software R (R Core Team 2012); the results included 95% bootstrap confidence intervals.

5.4 Results and discussion

5.4.1 One-factor-at-a-time

When the biofuel project has a low net GHG balance (scenario 1), the C3BO model results suggest that ethanol conversion efficiency and gasoline production emissions have the greatest impact on the GHG balance, followed by ethanol production emissions (Figure 5.1, Figure 5.3, and Figure 5.5). Biomass yield has a low impact on GHG balance.

[7] LMG is the R^2 contribution averaged over orderings among regressors (Lindeman *et al.* 1980)

[8] PMVD is the proportional marginal variance decomposition as proposed by Feldman (2005). It can be interpreted as a weighted average over orderings among regressors, with data-dependent weights

However, when the biofuel project has a high GHG balance (scenario 2), the results are different. The impact of biomass yield is large, similar to that of ethanol conversion efficiency. The affect of initial carbon stocks is also large.

It is interesting to note that in scenario 1, the affect of ethanol conversion efficiency becomes less important with time (*i.e.* from year 38 to 68, and to 100), while that of gasoline production emissions and of ethanol production emissions become more important with time. This suggests that, in the life cycle GHG of biofuels, it matters what kind of gasoline is being displaced by ethanol. For example in a 100-year project if one type of gasoline would have 10% less emissions then this would worsen the ethanol project net GHG balance by between 3.2-14.4%.

In scenario 2, the impact of biomass yield, ethanol conversion efficiency, gasoline production emissions, and initial carbon stocks becomes less important with time, while that of ethanol production emissions become more important.

Two important current areas of research are biomass improvements (yield) and conversion to ethanol efficiencies. Projects that have a large net GHG balance (scenario 2) will benefit from improvements in biomass yield (*i.e.* net GHG balance decreases as biomass yield increases; as previously mentioned in CHAPTER 4, a decrease in the net GHG balance of a biofuel project signifies that the biofuel production system produces less emissions that the fossil fuel system displaced, which is a desired outcome). The impact is reduced as project time horizon increases: a 10% improvement in biomass yield will result in a 6.7% reduction in net GHG38 balance, 5.3% reduction in GHG68, and 4.3% reduction in GHG100. However, our results suggest that improvements in biomass yield do not have a sizeable impact on projects with low net GHG balance (scenario 1). In other words, when project net emissions are small then biomass yield is not as important.

Improvements in ethanol conversion efficiency result in contrasting impacts. On one hand, projects with low GHG balance (scenario 1) are negatively affected by increases in ethanol conversion efficiency: a 10% improvement in conversion efficiency results in a 6.2% increase in net GHG38 balance, a 5.0% increase in GHG68, and 4.1% increase in GHG100 (note that an increase in the project net GHG balance is not a desirable outcome, as it means that the biofuel project produces more emissions than the displaced fossil fuel). This is explained by the fact that a key factor for a favourable GHG balance is the carbon sequestered in live biomass at the end of the project; the larger the plantation area, the larger the carbon sequestration. However, improvements in ethanol yield greatly reduce the number of hectares needed, from 96,356 ha to 87,596 ha (since the amount of ethanol produced annually is constant), thereby reducing the carbon sequestered in live biomass (from 112 to 102 kg CO_2/GJ EtOH sequestered at 38 years, from 73 to 66 kg CO_2/GJ EtOH at 68 years, and from 55 to 50 kg CO_2/GJ EtOH at 100 years), which was enough to worsen the GHG balance: from -95.8 to -88.5 kg CO_2/GJ EtOH net GHG balance at 38 years, from -60.3 to 56.2 kg CO_2/GJ EtOH at 68 years, and -44.7 to -42.0 kg CO_2/GJ EtOH at 100 years. In this case, the reduced GHG benefits of the project were due to the reduced afforestation. This means that the benefits were due to the afforestation, not to displacement of gasoline by ethanol.

On the other hand, projects with high GHG balance (scenario 2) are positively affected by improvements in conversion efficiency: a 10% improvement in conversion efficiency results in a 6.0% decrease in net GHG38 balance, a 5.3% decrease in GHG68, and 4.8% decrease in GHG100.

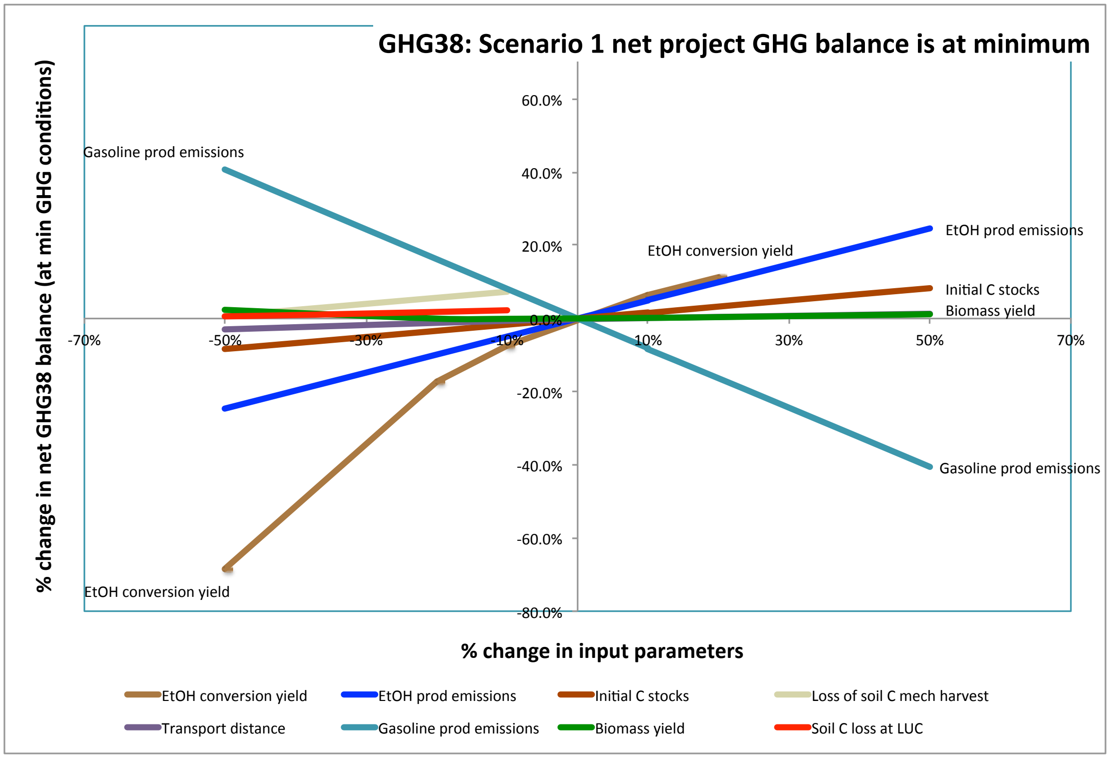

Figure 5.1. Influence of various input parameters on the net GHG38 balance, when project GHG balance is at minimum (scenario 1)

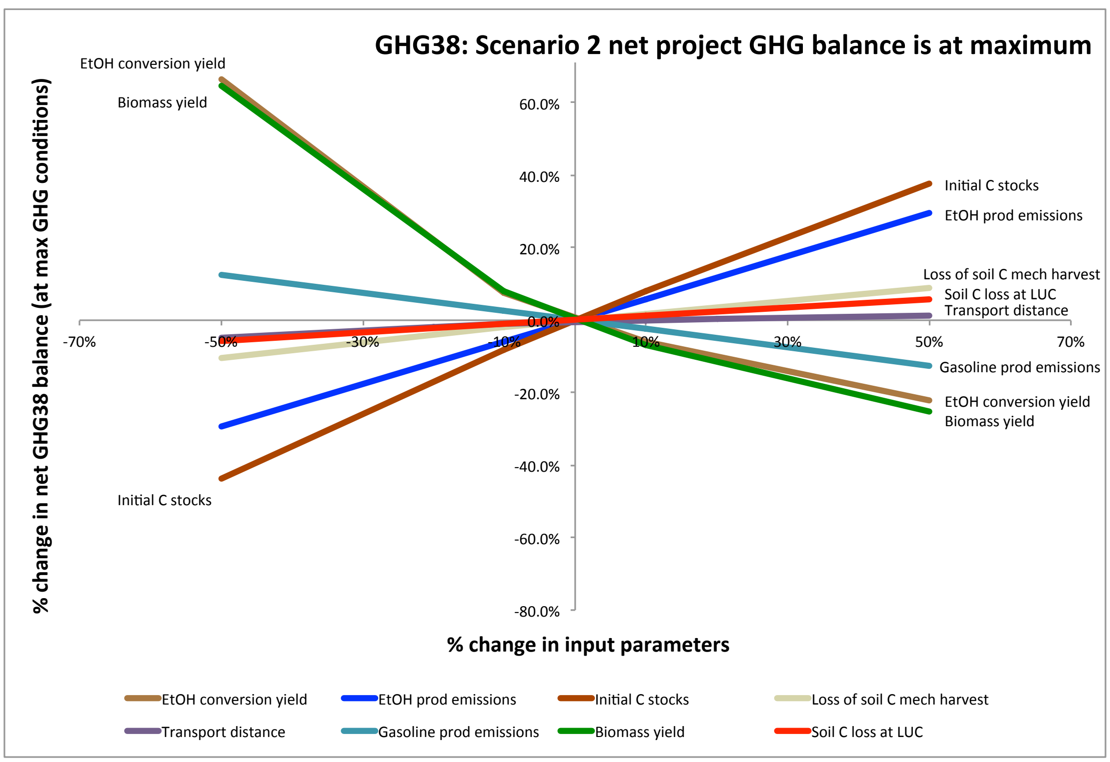

Figure 5.2. Influence of various input parameters on the net GHG38 balance, when project GHG balance is at maximum (scenario 2)

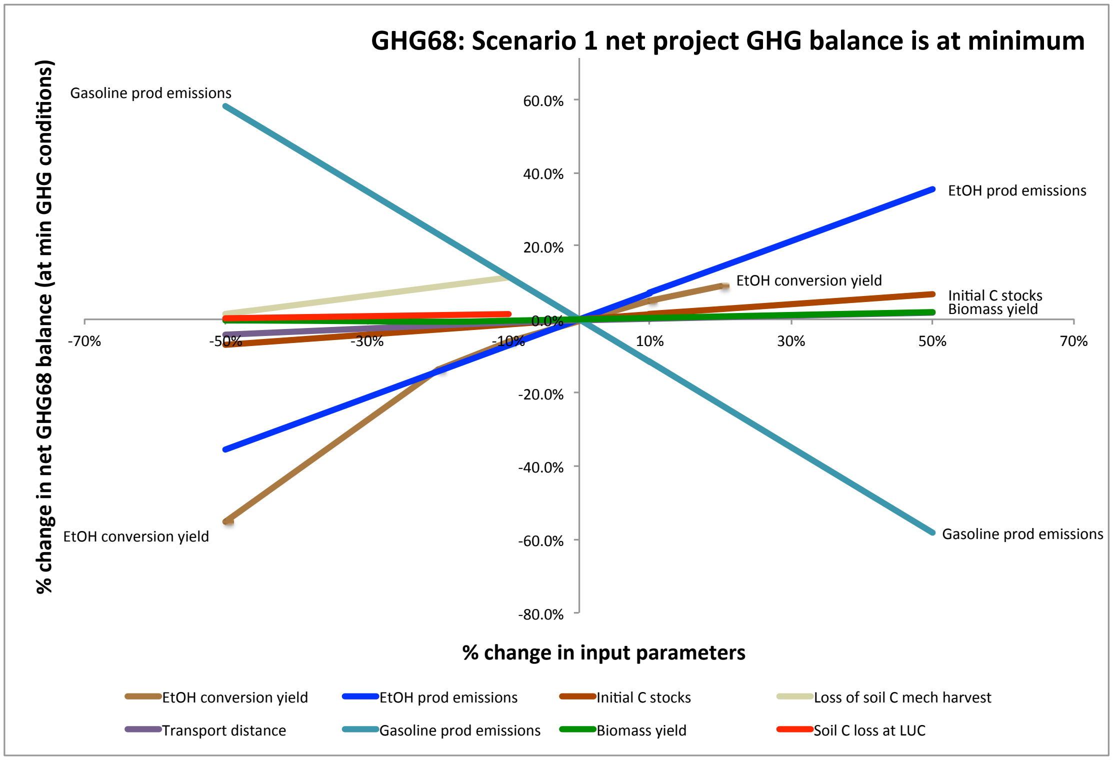

Figure 5.3. Influence of various input parameters on the net GHG68 balance, when project GHG balance is at minimum (scenario 1)

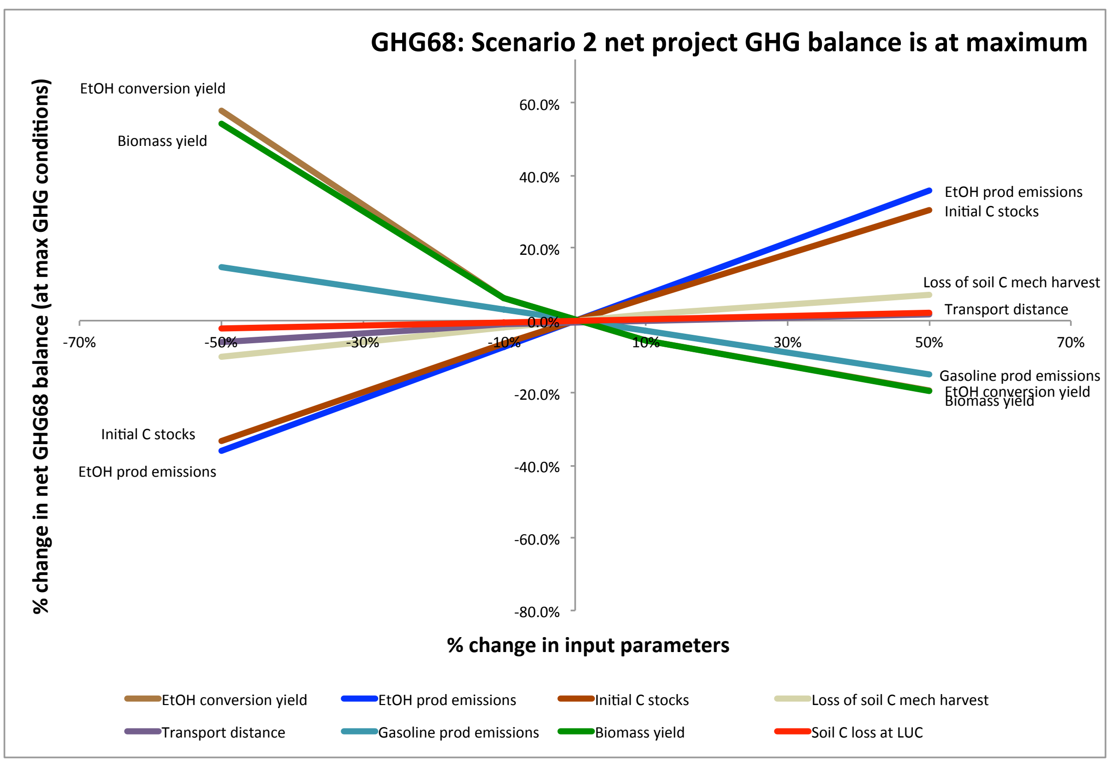

Figure 5.4. Influence of various input parameters on the net GHG68 balance, when project GHG balance is at maximum (scenario 2)

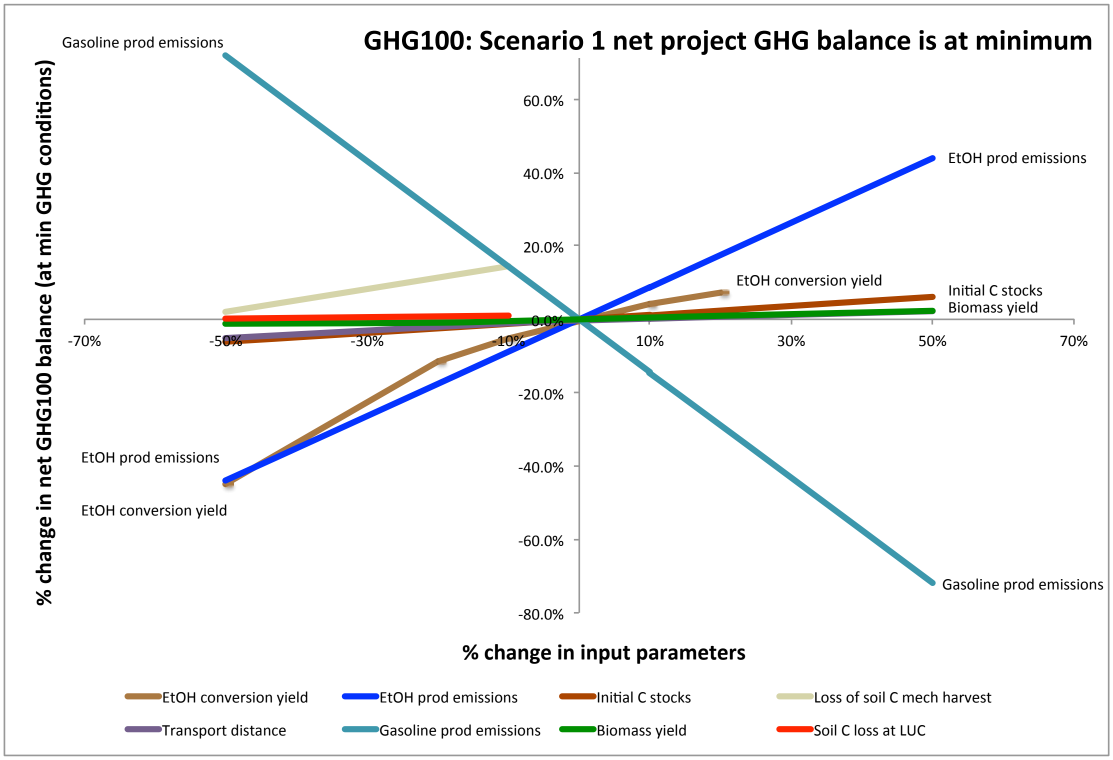

Figure 5.5. Influence of various input parameters on the net GHG100 balance, when project GHG balance is at maximum (scenario 1)

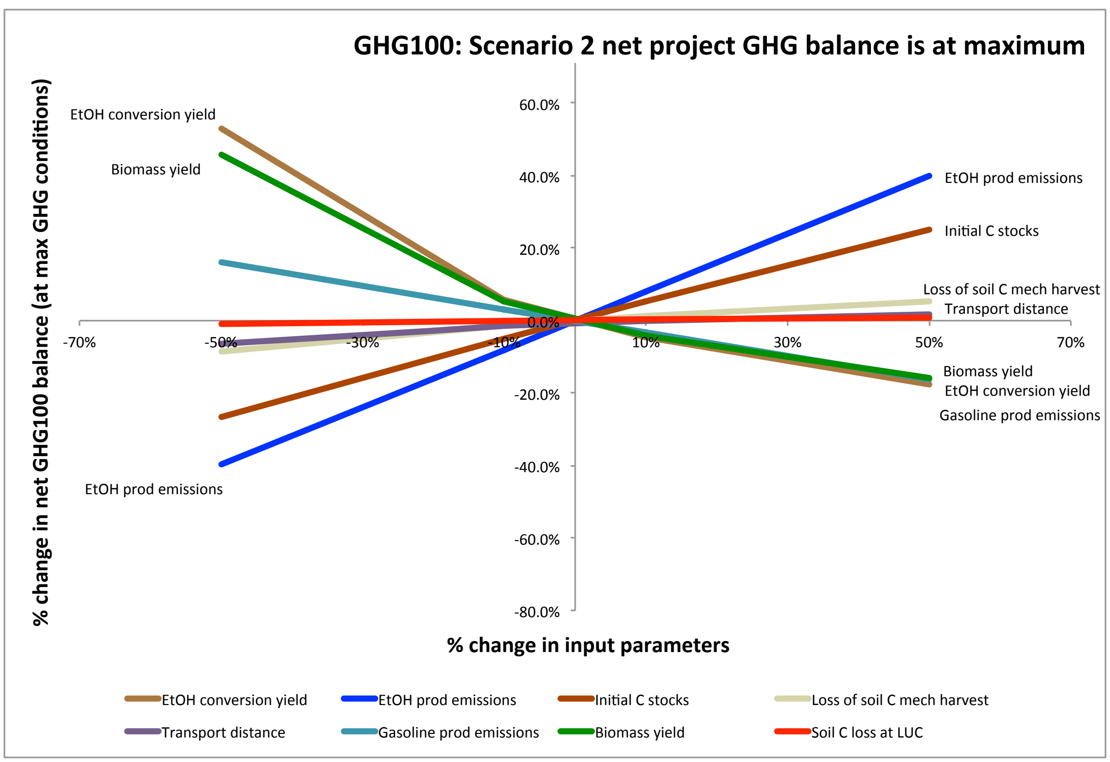

Figure 5.6. Influence of various input parameters on the net GHG100 balance, when project GHG balance is at maximum (scenario 2)

As shown in Figure 5.7 and Figure 5.8, the ethanol conversion efficiency has the most important impact on ethanol production cost, approximately double that of biomass conversion efficiency, for both scenarios 3 and 4. As expected, decreases in conversion efficiency and in biomass yield result in increases of ethanol cost. Likewise, increases in conversion and biomass yields lead to decreases in ethanol cost.

A 10% improvement in ethanol conversion efficiency results in a 5.3-5.7% reduction in ethanol production cost, while a 10% improvement in biomass yield reduces ethanol cost by 2.1-2.2%. This compares with Sassner *et al.* (2008) who reported a 10% increase in ethanol yield would reduce the production cost by almost 7%.

The plantation area is impacted in (proportionally) comparable ways by biomass yield, ethanol conversion efficiency, and biorefinery capacity (Figure 5.9 and Figure 5.10). As expected, increases in biomass and conversion yield reduce the plantation area needed, and increases in biorefinery capacity increase the area. A 10% improvement in either biomass yield or conversion efficiency results in a 9.1% decrease in plantation area.

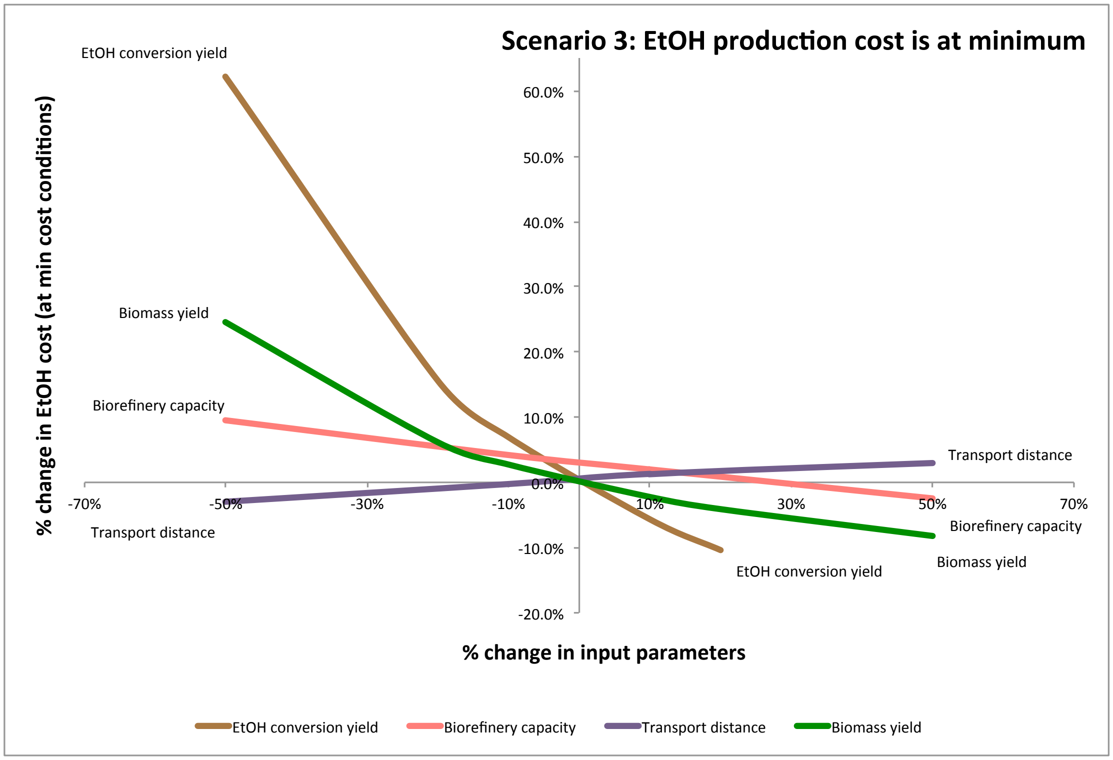

Figure 5.7. Influence of various input parameters on ethanol production cost, when ethanol cost is at minimum (scenario 3)

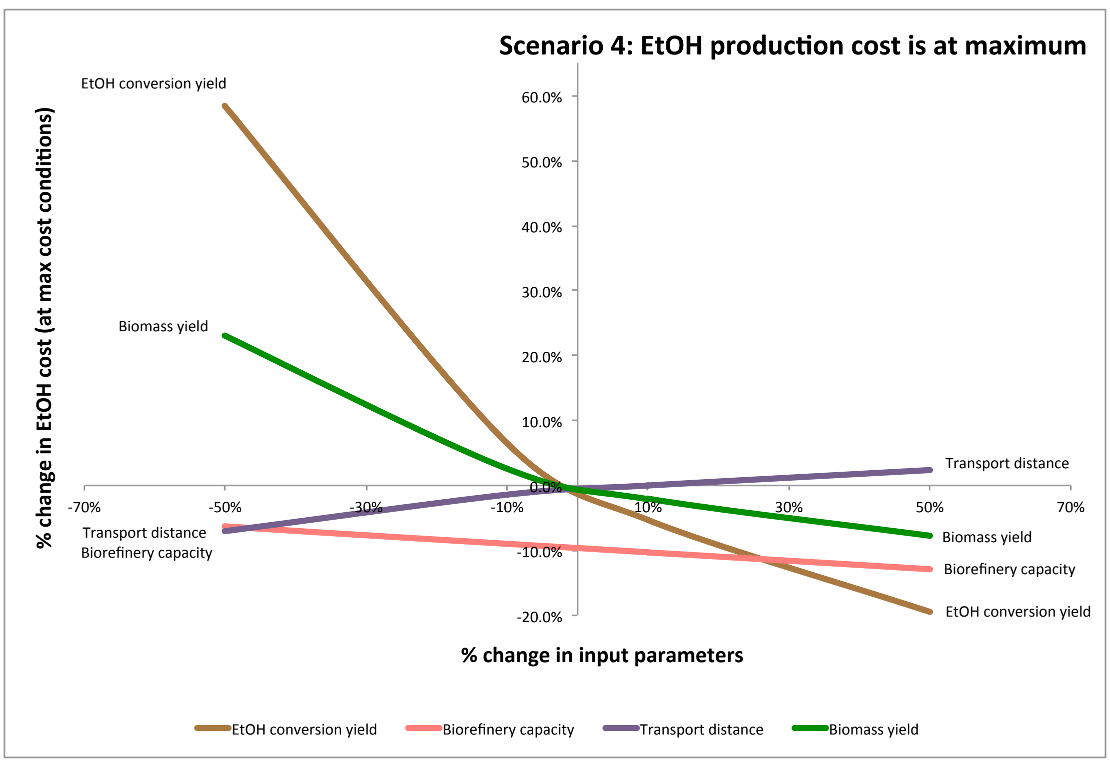

Figure 5.8. Influence of various input parameters on ethanol production cost, when ethanol cost is at maximum (scenario 4)

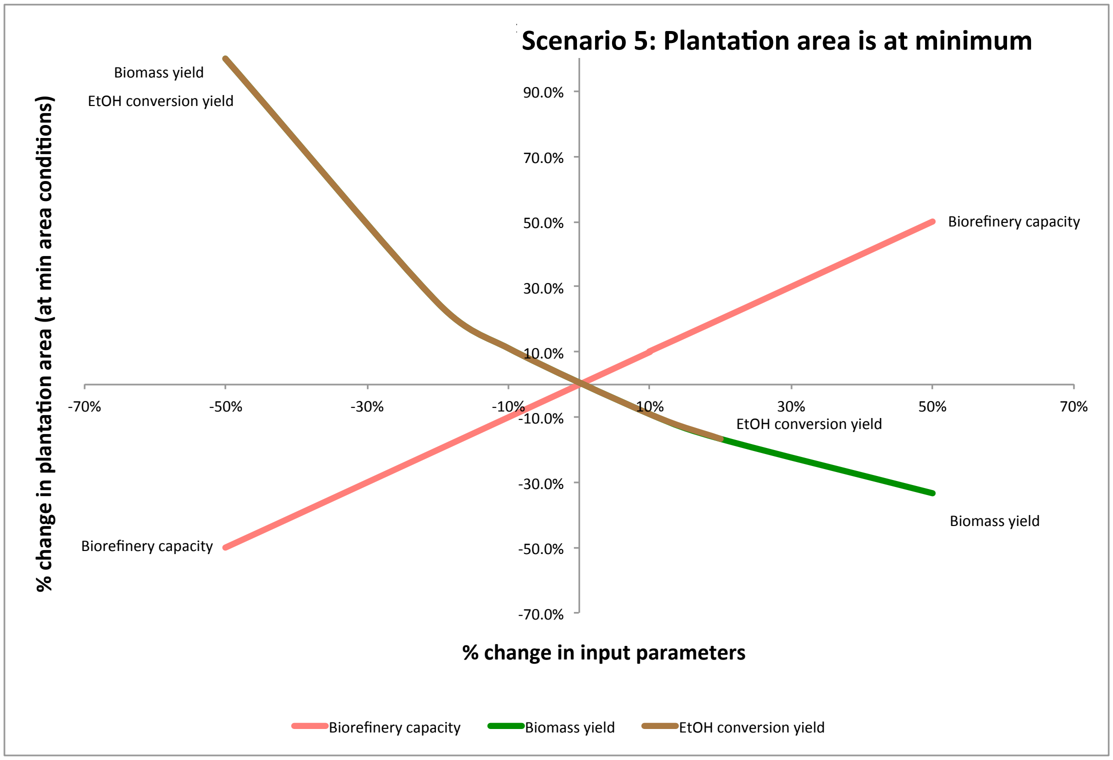

Figure 5.9. Influence of various input parameters on plantation area, when area is at minimum (scenario 5)

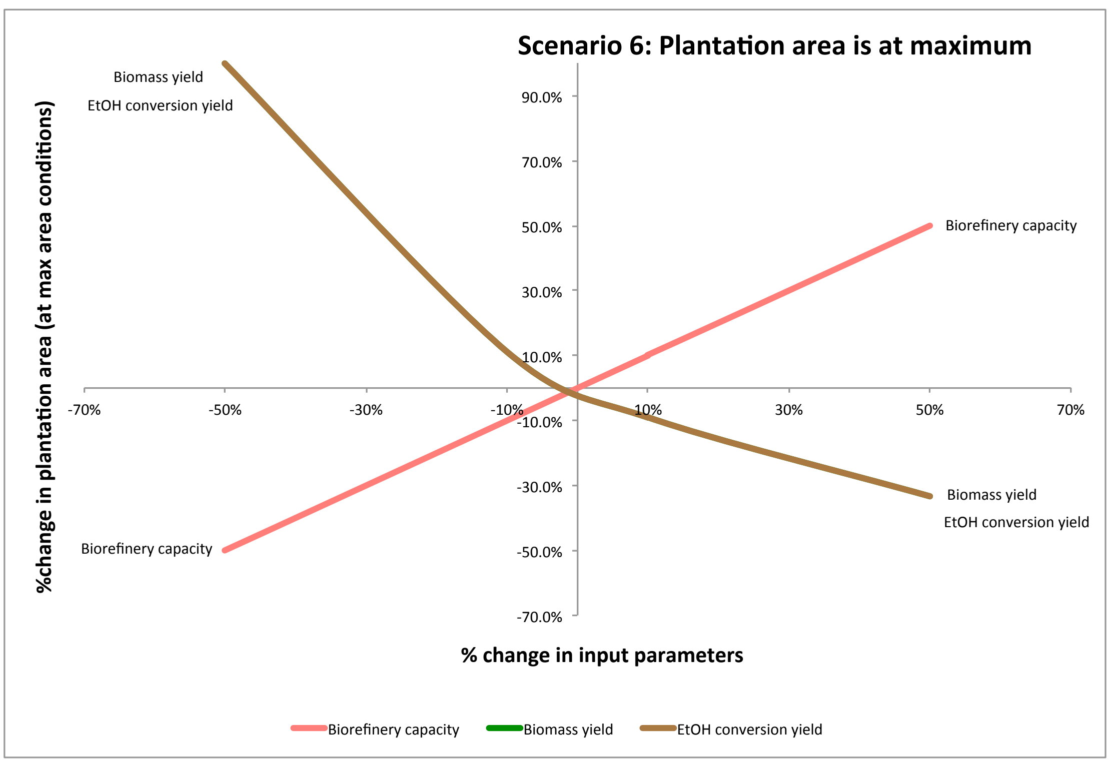

Figure 5.10. Influence of various input parameters on plantation area, when area is at maximum (scenario 6)

5.4.2 Multiple linear regression

The sensitivity analysis using multiple linear regressions was conducted separately for the ethanol production cost, plantation area, and project net GHG balance.

5.4.2.1 Dependent variable: ethanol cost

The AIC stepwise regression procedure kept 4 of the initial 5 variables in the final model: v01 biomass yield, v02 transportation distance, v03 conversion efficiency, and v04 biorefinery capacity.

Table 5.2. Changes in net GHG38 balance

		Change in input parameters					
		-50%	*-20%*	*-10%*	*+10%*	*+20%*	*+50%*
		Change in GHG38 balance					
EtOH prod emissions	minGHG	-24.6%		-4.9%	4.9%		24.6%
	maxGHG	-29.4%		-5.9%	5.9%		29.4%
Initial C stocks	minGHG	-8.5%		-1.7%	1.7%		8.3%
	maxGHG	-43.9%		-8.2%	8.0%		37.7%
Loss of soil C mech harvest	minGHG	--		--	0.7%*		7.3%*
	maxGHG	-10.5%		-2.0%	1.9%		9.1%
Transport distance	minGHG	-3.0%		-0.6%	0.1%		1.3%
	maxGHG	-4.8%		-1.0%	0.1%		1.4%
Gasoline prod emissions	minGHG	40.6%		8.1%	-8.1%		-40.6%
	maxGHG	12.6%		2.5%	-2.5%		-12.6%
Biomass yield	minGHG	2.3%	0.0%	-0.1%	0.2%	0.4%	1.2%
	maxGHG	64.6%		8.0%	-6.7%		-25.2%
Soil C loss at LUC	minGHG	--		--	0.5%§		2.3%§
	maxGHG	-5.7%		-1.1%	1.1%		5.7%
EtOH conversion efficiency	minGHG	-68.5%	-17.1%	-7.6%	6.2%	11.4%	--
	maxGHG	66.4%		7.4%	-6.0%		-22.1%

* increases are to 1% and to mid-range of loss of soil carbon due to mechanized activities at harvest, respectively; effect of decreases not calculated since minimum of range is zero

§ increases to are 2.5% and to mid-range of soil carbon loss at land-use change, respectively; effect of decreases not calculated since minimum of range is zero

Note: negative sign results signify decreases in net balance GHG emissions.

An analysis of variance showed highly significant effects for all factors: biomass yield $F(1,40) = 13.66$, $p < 0.001$; transportation distance $F(1,40) = 149.52$, $p < 0.001$; conversion efficiency $F(1,40) = 844.87$, $p < 0.001$; biorefinery capacity $F(1,40) = 181.31$, $p < 0.001$. The ANOVA results are shown in Table 5.3.

Table 5.3. ANOVA table for the chosen model for Ethanol Cost

ANOVA EtOH Cost	Df	Sum Sq	Mean Sq	F value	Pr(>F)	
Biomass yield; v01	1	0.00369	0.00369	13.66	0.000657	***
Transp. dist.; v02	1	0.04035	0.04035	149.52	4.32E-15	***
Conversion efficiency; v03	1	0.22801	0.22801	844.87	<2.00E-16	***
Biorefinery capacity; v04	1	0.04893	0.04893	181.31	<2.00E-16	***
Residuals	40	0.0108	0.00027			
Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1						

A multiple linear regression analysis was conducted to determine if ethanol cost could be predicted from the 4 variables selected, and the results are shown in Table 5.4. The null hypothesis (that the regression coefficient, *i.e.* the slope, was equal to 0) was rejected, $F(4, 40) = 297.3$, $p = < 0.001$. The squared multiple correlation R^2=0.9675 indicates that 96.75% of the variability in the EtOH cost is explained by the 4 input variables.

Table 5.4. Linear regression model for dependent variable Ethanol Cost

	Estimate	Std. Error	t value	Pr(>|t|)	
(Intercept)	1.989	4.54E-02	43.778	<2.00E-16	***
Biomass yield; v01	-0.012	1.77E-03	-6.888	1.52E-08	***
Transp. dist.; v02	6.52E-04	5.99E-05	10.884	5.53E-14	***
Conversion efficiency; v03	-3.71E-03	1.27E-04	-29.306	<2.00E-16	***
Biorefinery capacity; v04	-3.78E-04	2.81E-05	-13.465	<2.00E-16	***
Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					
Residual standard error: 0.01643 on 40 degrees of freedom					
Multiple R-squared: 0.9675, Adjusted R-squared: 0.9642					
F-statistic: 297.3 on 4 and 40 DF, p-value: < 2.2e-16					

The data were screened for completeness and violation of assumptions prior to analysis. There was no missing data. Linearity and homoscedasticity were tested using Residuals *vs*. Fitted and Scale-Location plots, Breusch-Pagan test against heteroscedasticity ($p = 0.74$; fail to reject H0 that data is heteroscedastic, *i.e.* there is no evidence that the data is not homoscedastic), and Ramsey's RESET test of functional misspecification ($P = 0.74$; fail to reject the null hypothesis of no misspecification). The large coefficient of determination (R^2=0.97) also suggests linearity. After the Shapiro-Wilk test for normality of residuals, $W = 0.96$, $p = 0.08$, we fail to reject the null that the residuals are normally distributed at an alpha level of 0.05.

The linear regression model for ethanol cost is shown in equation 5.1:

$$\text{EtOH Cost} = 1.989 + v01 * -0.012 + v02 * 6.52\text{E-}04 + v03 * -3.71\text{E-}03 + v04 * -3.78\text{E-}04 \tag{5.1}$$

The LMG and PMVD metrics shown in Table 5.5 can be used as direct measures of sensitivity. The results suggest that the EtOH regression model is most sensitive to conversion efficiency (LMG = .742; PVMD = .765), and much less sensitive to the other input variables, as shown in Figure 5.11.

These results are somewhat similar to those from the OFAT analysis described earlier. However, it is evident that the multiple regression analysis that includes model results across the entire range of possible ethanol costs (compared with only minimum or only maximum costs in OFAT scenarios 3 and 4 above) is unable to reflect the non-linear behaviour of the impact of biomass yield and conversion efficiency (which can be observed in Figure 5.7 and Figure 5.8 above). Another reason why biorefinery capacity and transportation distance have a larger impact than biomass yield is the fact that the former two variables' variation within their ranges is much larger: the range of possible values for biorefinery capacity and transportation distance is ±67% for both, while the range for biomass yield is much smaller, ±18%.

Table 5.5. LMG metrics for relative importance of regressor variables to multiple regression models for three dependent variables

	Dependent variables					
	EtOHcost		Area		Net GHG balance 100 years	
Regressor variables	*LMG*	*PMVD*	*LMG*	*PMVD*	*LMG*	*PMVD*
Biomass MAI v01	.017	.030	.061	.045	.008	.005
Biomass transportation distance v02	.111	.078			.015	.011
Conversion efficiency v03	.718	.740	.018	.017	.003	.002
Biorefinery capacity v04	.122	.119	.921	.938		
Initial carbon stocks v06					.425	.416
Loss of soil C (slow decaying pools) from mechanized activities in harvest years v07					.020	.044
EtOH production emissions v08					.513	.502
Gasoline production emissions v09					.012	.019
Soil carbon loss at land-use change v10					.006	.001

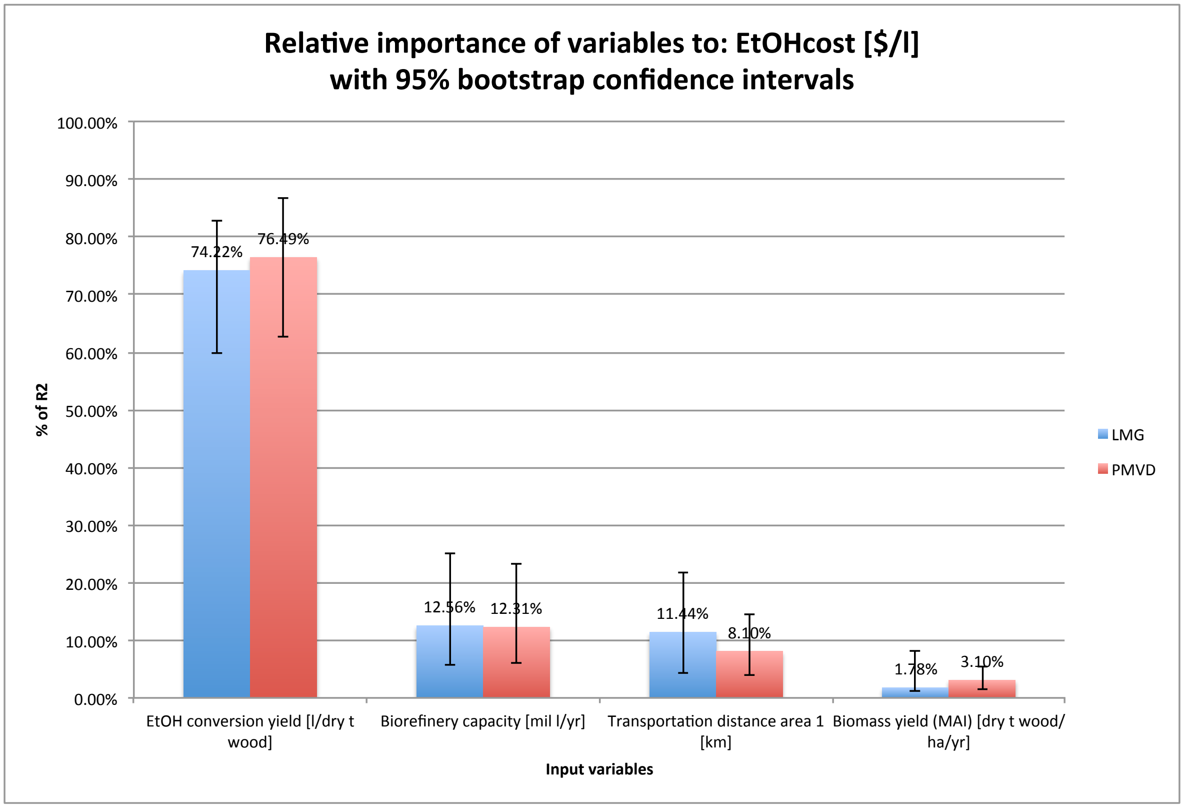

Figure 5.11. Relative importance of input variables to the multiple regression model of total production cost of ethanol

5.4.2.2 Dependent variable: plantation area

The stepwise regression procedure kept 3 of the initial 7 variables in the final model: i) biomass yield, ii) conversion efficiency, and iii) biorefinery capacity.

An analysis of variance showed that there were significant effects for: biomass yield $F(1,41) = 230$, $p < 0.001$; conversion efficiency $F(1,41) = 108.2$, $p < 0.001$; biorefinery capacity $F(1,41) = 3074.6$, $p < 0.001$. The ANOVA table is shown in Table H.1 in APPENDIX H.

Linearity and homoscedasticity were tested using Residuals *vs*. Fitted and Scale-Location plots, Breusch-Pagan test against heteroscedasticity (p = 0.06; fail to reject H0 that data is heteroscedastic), and Ramsey's RESET test of functional misspecification ($p = 0.012$; we reject the null hypothesis of no misspecification, at the 0.05 alpha level). Ramsey's RESET test indicates that the linearity of the regression model for plantation area assumption is not reasonable. The Residuals *vs*. Fitted plot exhibited possible non-linearity. The Q-Q plot shows some departure from normality, and the Shapiro-Wilk test for normality ($p = 0.04$) suggests that normality assumption of residuals is not reasonable.

A multiple linear regression analysis was conducted to determine if plantation area could be predicted from the 3 variables selected, and the results are shown in Table H.2 in APPENDIX H. The null hypothesis (that the regression coefficient, *i.e.* the slope, was equal to 0) was rejected, $F(3, 41) = 1138$, $p = < 0.001$. The squared multiple correlation R^2=0.9881, indicating that 98.81% of the variability in the area variable is explained by the 3 input variables.

The linear regression model for plantation area is shown in equation 5.2:

$$\text{Plantation Area} = 130703.09 + v01 * -5103.47 + v03 * -235.58 + v04 * 343.50 \quad (5.2)$$

The LMG and PMVD metrics shown in Table 5.5 suggest that the area regression model is most sensitive to biorefinery capacity (LMG = .921; PVMD = .938), and much less sensitive to the other input variables (Figure 5.12). This result might seem contradictory to the results of scenarios 3 and 4, where all three input variables have similar impacts on ethanol cost. However, this is explained by the fact that the range of possible values for biorefinery capacity is much larger (±67%) than the range of biomass yield (±18%) and conversion efficiency (±13%).

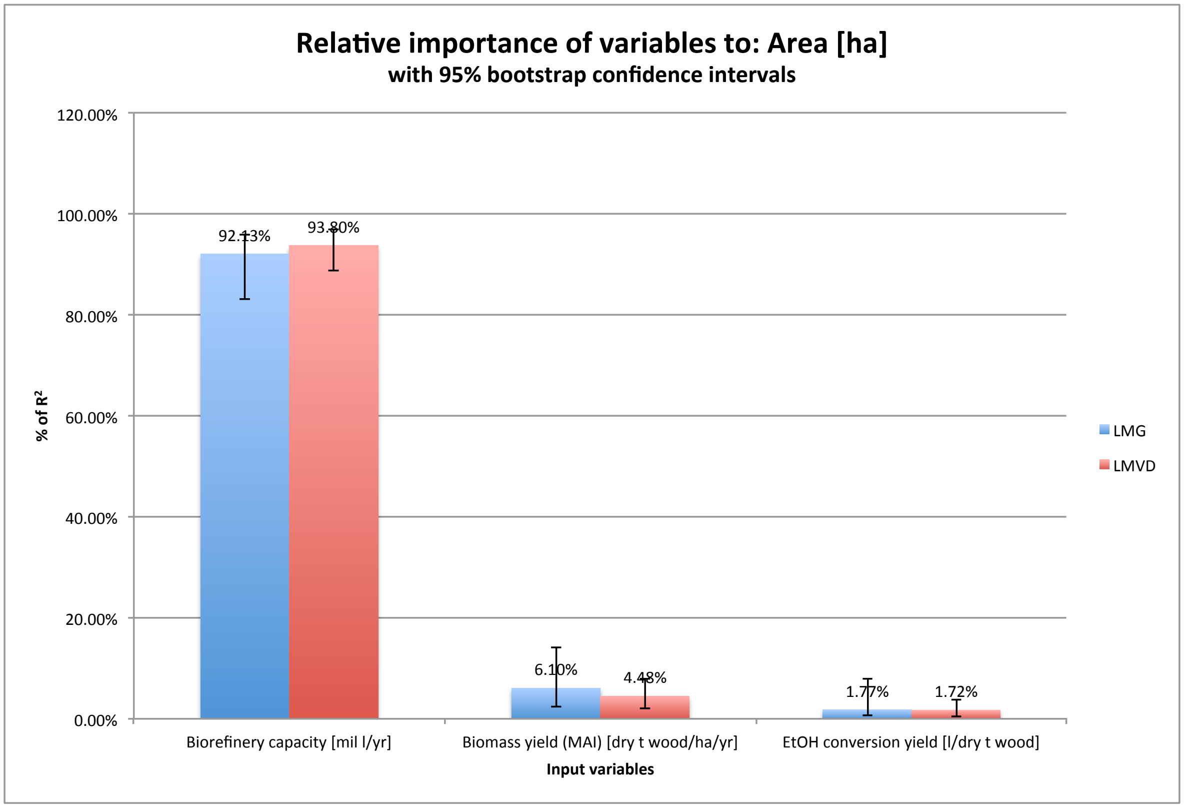

Figure 5.12. Relative importance of input variables to the multiple regression model of plantation area

5.4.2.3 Dependent variable: project net GHG balance

Dependent variable: net GHG balance at year 38

The stepwise regression procedure kept 8 of the initial 13 variables in the final model (Table H.3 in APPENDIX H).

An analysis of variance showed that there were significant effects for: biomass yield $F(1,36) = 43.0$, $p < 0.001$; initial carbon stocks $F(1,36) = 1715.7$, $p < 0.001$; loss of soil C from mechanized activities in harvest years $F(1,36) = 42.2$, $p < 0.001$; ethanol production emissions $F(1,36) = 355.6$, $p < 0.001$; gasoline production emissions $F(1,36) = 11.4$, $p = 0.002$; carbon soil loss at land-use change $F(1,36) = 15.2$, $p < 0.001$. The ANOVA table is shown in Table H.3 in APPENDIX H.

A multiple linear regression analysis was conducted to determine if the net GHG balance 38 years could be predicted from the 8 variables selected, and the results are shown in Table H.4 in APPENDIX H. The null hypothesis (that the regression coefficient, *i.e.* the slope, was equal to 0) was rejected, $F(8, 36) = 273.3$, $p = < 0.001$. The squared multiple correlation $R^2=0.9838$, indicating that 98.38% of the variability in the GHG balance 38 years variable is explained by the 9 input variables.

Linearity and homoscedasticity were tested using Residuals *vs*. Fitted and Scale-Location plots, Breusch-Pagan test against heteroscedasticity ($p = 0.23$; fail to reject H0 that data is heteroscedastic), and Ramsey's RESET test of functional misspecification ($P = 0.34$; fail to reject the null hypothesis of no misspecification). The large coefficient of determination ($R^2=0.\ 9838$) also suggests linearity. The Q-Q plot supports the assumption of normality of residuals, and Shapiro-Wilk test for normality ($p = 0.93$) fails to reject H0 that residuals are normally distributed.

The linear regression model for net GHG balance year 38 is shown in equation 5.3:

$$\text{Net GHG balance year 38} = -20.26 + v01 * -6.31 + v06 * 4.08 + v06 * 515.99 + v02 * 0.15 + v03 * -0.12 + v08 * 138.21 + v09 * -103.71 + v10 * 89.95 \quad (5.3)$$

The LMG and PMVD metrics shown in Table 5.5 suggest that the regression model for the net GHG balance 38 is most sensitive to initial carbon stocks (LMG = .738; PVMD = .734), less sensitive to EtOH production emissions (LMG = .211; PVMD = .236), and much less sensitive to the other input variables, as shown in Figure 5.13.

Again, at first these results may seem to be at odds with the results of scenarios 1 and 2 above, where ethanol conversion efficiency, biomass yield, ethanol production emissions, and gasoline production emissions also had an important impact on the GHG balance. However, this is explained by the fact that the initial carbon stocks variation within its range of possible values was larger (±71%) than ethanol production emissions (±52%), biomass yield (±18%), gasoline production emissions (±13%), and ethanol conversion efficiency (±13%).

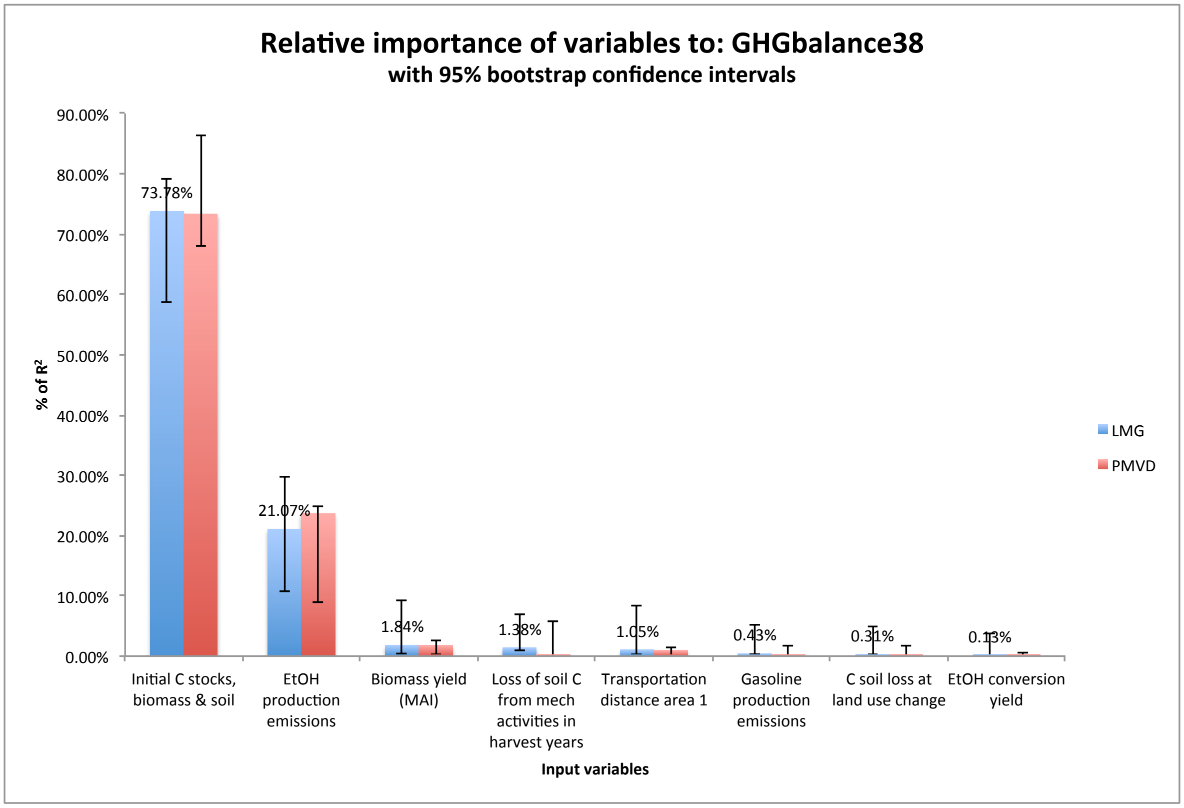

Figure 5.13. Relative importance of input variables to the multiple regression model of net GHG balance at 38 years

Dependent variable: net GHG balance at year 68

The stepwise regression procedure kept 8 of the initial 12 variables in the final model (Table H.5 in APPENDIX H).

An analysis of variance showed that there were significant effects for: biomass yield $F(1,36) = 25.2$, $p < 0.001$; initial carbon stocks $F(1,36) = 1523.1$, $p < 0.001$; loss of soil C activities in harvest years $F(1,36) = 54.9$, $p < 0.001$; transportation distance $F(1,36) = 5.4$, $p = 0.026$; ethanol production emissions $F(1,36) = 833.7$, $p < 0.001$; gasoline production emissions $F(1,36) = 25.5$, $p < 0.001$; carbon soil loss at land-use change $F(1,36) = 5.8$, $p = 0.02$. The ANOVA table is shown in Table H.5 in APPENDIX H.

A multiple linear regression analysis was conducted to determine if the net GHG balance 68 years could be predicted from the 8 variables selected, and the results are shown in Table H.6 in APPENDIX H. The null hypothesis (that the regression coefficient, *i.e.* the slope, was equal to 0) was rejected, $F(8, 36) = 309.5$, $p = < 0.001$. The squared multiple correlation R^2=0.9857, indicating that 98.57% of the variability in the GHG balance 68 years variable is explained by the 9 input variables.

Linearity and homoscedasticity were tested using Residuals *vs*. Fitted and Scale-Location plots, Breusch-Pagan test against heteroscedasticity (p = 0.30; fail to reject H0 that data is heteroscedastic), and Ramsey's RESET test of functional misspecification (P = 0.29; fail to reject the null hypothesis of no misspecification). The large coefficient of determination (R^2=0. 9857) also suggests linearity. The Q-Q plot indicates some departure from normality in the tails, however, the Shapiro-Wilk test for normality (*p* = 0.54) fails to reject H0 that residuals are normally distributed.

The linear regression model for net GHG balance year 68 is shown in equation 5.4:

Net GHG balance year 68 = 5.15 + v01 * -4.16 + v06 * 2.58 + v07 * 445.67 + v02 * 0.14 + v03 * -0.12 + v08 * 145.31 + v09 * -105.31 + v10 * 38.76   5.4

The LMG and PMVD metrics shown in Table 5.5 suggest that the regression model for the net GHG balance 68 is most sensitive to initial carbon stocks (LMG = .557; PVMD = .568), somewhat less sensitive to EtOH production emissions (LMG = .385; PVMD = .398), and much less sensitive to the other input variables (Figure 5.14).

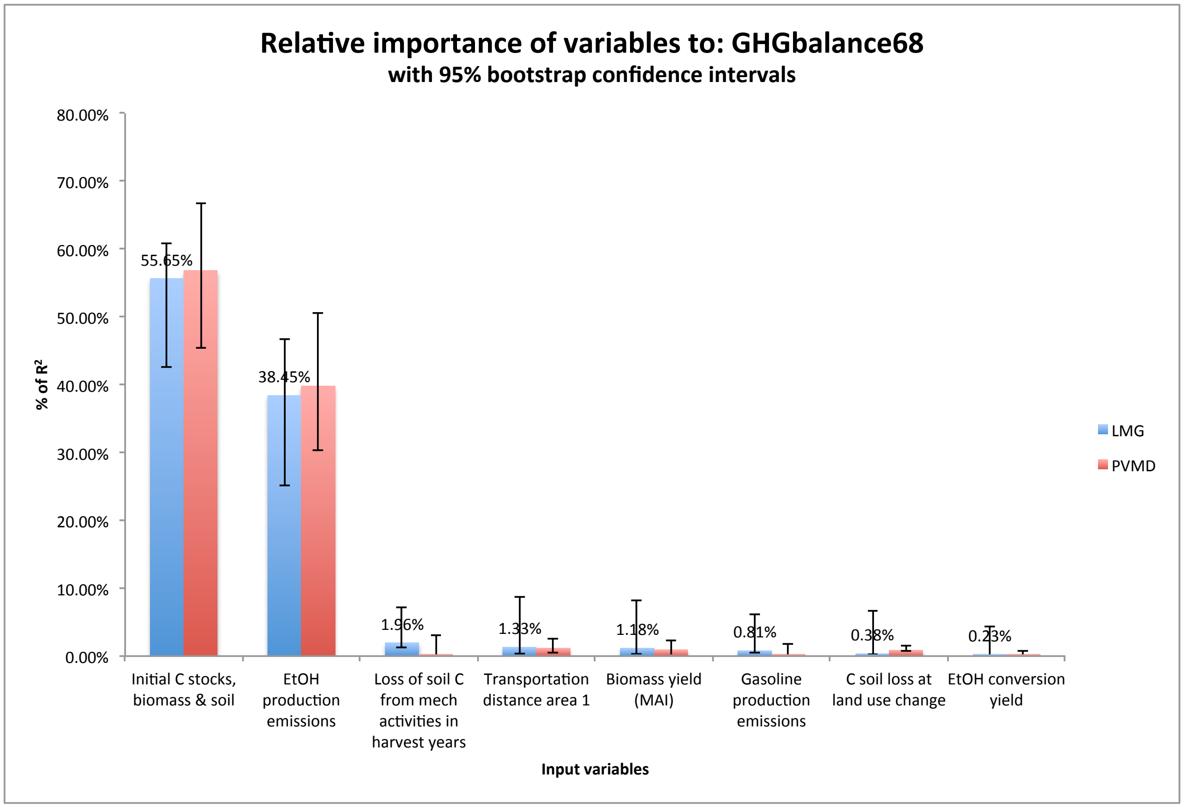

Figure 5.14. Relative importance of input variables to the multiple regression model of net GHG balance at 68 years

Dependent variable: net GHG balance at year 100

The stepwise regression procedure kept 8 of the initial 12 variables in the final model (Table H.7 in APPENDIX H).

An analysis of variance showed that there were significant effects for: biomass yield $F(1,36) = 15.3$, $p < 0.001$; initial carbon stocks $F(1,36) = 1508.8$, $p < 0.001$; loss of soil C from mechanized activities in harvest years $F(1,36) = 58.3$, $p < 0.001$; transportation distance $F(1,36) = 8.9$, $p = 0.005$; ethanol production emissions $F(1,36) = 1459.8$, $p < 0.001$; gasoline production emissions $F(1,36) = 44.0$, $p < 0.001$. The ANOVA table is shown in Table H.7 in APPENDIX H.

A multiple linear regression analysis was conducted to determine if the net GHG balance 100 years could be predicted from the 8 variables selected, and the results are shown in Table H.8 in APPENDIX H. The null hypothesis (that the regression coefficient, *i.e.* the slope, was equal to 0) was rejected, $F(8, 36) = 387.7$, $p = < 0.001$. The squared multiple correlation R^2=0.9885, indicating that 98.85% of the variability in the GHG balance 100 years variable is explained by the 9 input variables.

Linearity and homoscedasticity were tested using Residuals *vs*. Fitted and Scale-Location plots, Breusch-Pagan test against heteroscedasticity ($p = 0.38$; fail to reject H0 that data is heteroscedastic), and Ramsey's RESET test of functional misspecification ($P = 0.38$; fail to reject the null hypothesis of no misspecification). The large coefficient of determination (R^2=0. 9885) also suggests linearity. The Q-Q plot indicates some departure from normality in the tails; however, the Shapiro-Wilk test for normality ($p = 0.31$) fails to reject H0 that residuals are normally distributed.

The linear regression model for net GHG balance year 100 is shown in equation 5.5:

$$\text{Net GHG balance year 100} = 16.01 + v01 * -3.11 + v06 * 1.91 + v07 * 385.53 + v02 * 0.14 + v03 * -0.12 + v08 * 148.17 + v09 * -106.07 + v11 * 20.75 \quad (5.5)$$

The LMG and PMVD metrics shown in Table 5.5 suggest that the regression model for the net GHG balance 100 is most sensitive to EtOH production emissions (LMG = .513; PVMD = .514) and initial carbon stocks (LMG = .425; PVMD = .447), and much less sensitive to the other input variables (Figure 5.15). The net GHG balance 100 years regression model is very little sensitive to biomass yield or to conversion efficiency.

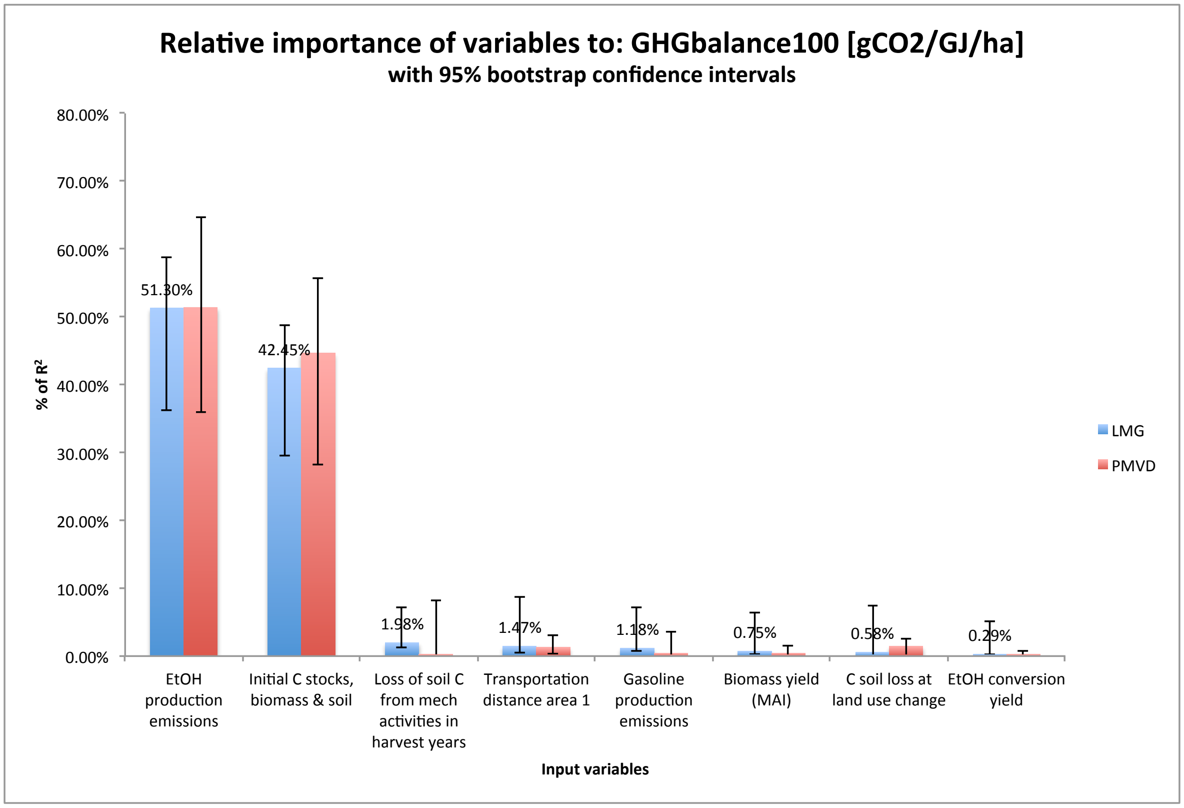

Figure 5.15. Relative importance of input variables to the multiple regression model of net GHG balance at 100 years

5.4.2.4 Dynamics of the relative importance of input variables to the net GHG balance by project time horizon

An interesting (however, not unexpected) result from the multiple linear regression sensitivity analysis is that the relative importance of some of the input variables changes with time, *i.e.* is different for different time horizons. As shown in Figure 5.16, the relative importance of initial carbon stocks to the project net GHG balance decreases with time, while the impact of ethanol production emissions increases with time. This is a direct consequence of the timing of the potential impacts at land-use change on the initial carbon stocks. For short projects, the initial carbon impacts due to land-use change are larger compared with all other project emissions and sequestration. As projects become longer, the one-time carbon impacts at land-use change represent a lower proportion of the other GHG emissions in the project. This indicates that the results of life cycle analyses for biofuel GHG can be affected by the choice of project time horizon.

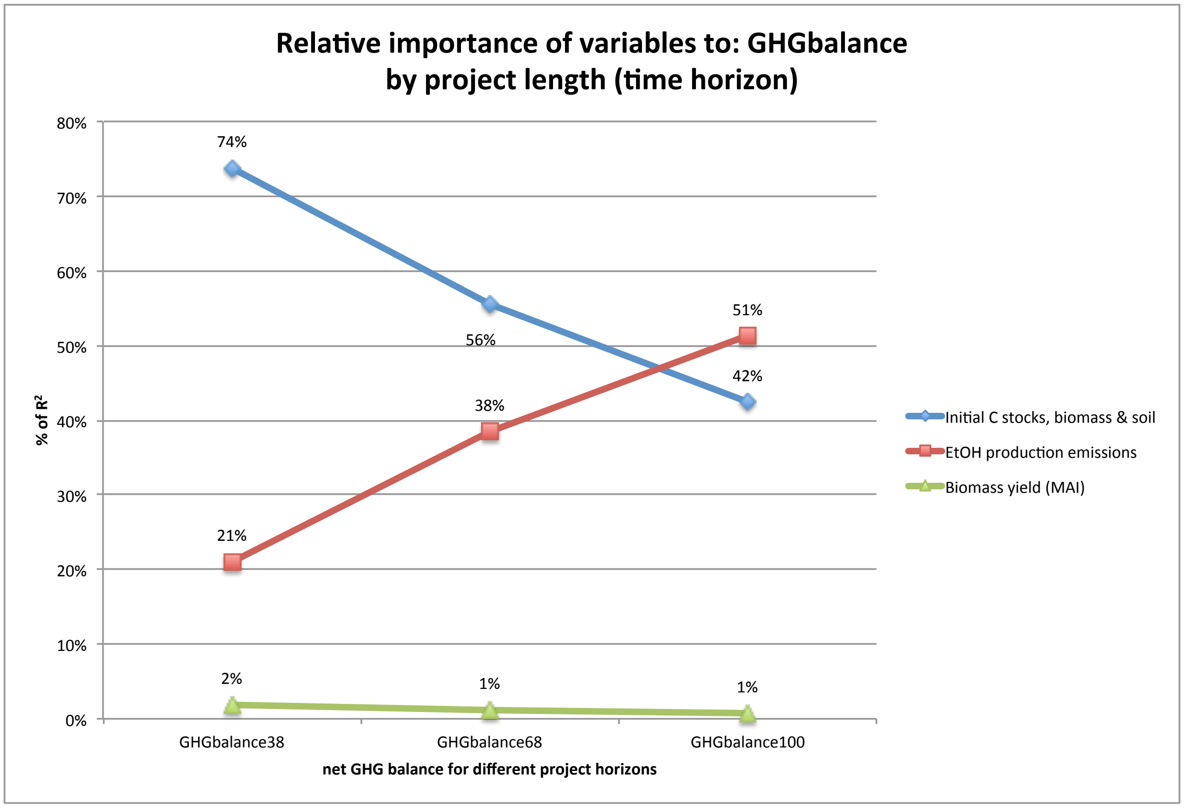

Figure 5.16. The relative importance of input variables to the net GHG balance, for different project time horizons

5.5 Limitations of the study

The sensitivity analysis methods employed in this chapter have inherent limitations that are common to all sensitivity analyses: 1) most sensitivity analysis methods assume independence between model inputs, but sometimes inputs can be strongly correlated; 2) the multiple linear regression approach can inaccurately measure sensitivity if the model response is nonlinear with respect to its input (however, our tests of the linearity assumption found it to be reasonable); 3) the one-factor-at-a-time approach does not fully explore the input space, since it does not take into account the simultaneous variation of input variables, or detect the presence of interactions between input variables (Czitrom 1999); however, when interactions between regression factors have been added to the regression model, none were found to be significant).

The sensitivity analysis based on multiple linear regression can be sensitive to the range of variation in the data used to fit the model, and may not always clearly reveal a relationship that actually exists (Frey and Patil 2002).

5.6 Conclusion

For the input variables considered in this study, the sensitivity analysis using linear regression indicates that the net GHG balance is most sensitive to ethanol production emissions and initial carbon stocks, and much less sensitive to other input variables. Biomass yield and conversion efficiency each explain less than 2% of the variability in net GHG balance at 38, 68 and 100 years, across the range of input variables values considered. As the project time horizon increases, the impact of initial carbon stocks decreases, and that of ethanol production emissions increases.

When a biofuel project has a very low net GHG balance, changes in ethanol conversion efficiency and in gasoline production emissions have the most impact on the GHG balance, followed by ethanol production emissions. *Ceteris paribus*, a 10% improvement in conversion efficiency results in a 4.1% – 6.2% increase in net GHG balance, depending on the project time horizon. Although this result seems counterintuitive, the increase in GHG balance is explained by the fact that increased conversion efficiency results in less plantation area and therefore less carbon sequestered in live biomass and soil at the end of the project. The carbon benefits of the project are related to afforestation, and therefore less plantation area will sequester less carbon. This indicates that improvements in conversion efficiency are not beneficial to the GHG balance of low emissions projects.

A 10% change in ethanol production emissions of low GHG projects results in a 4.9% – 8.8% change in net GHG balance, and a 10% change in gasoline production emissions results in a 8.1% – 14.4% change in net GHG balance. Biomass yield and initial carbon stocks have a small impact on the GHG balance for projects that have a low net GHG balance. This suggests that improvements in biomass yield do not bring important benefits to the GHG balance of low emissions projects.

However, when a biofuel project has a very high GHG balance, the impact of changes in biomass yield is large, and is similar to that of ethanol conversion efficiency. A 10%

improvement in biomass yield would result in a reduction in net GHG balance of 4.3% – 6.7%, depending on the project time horizon. A 10% improvement in conversion efficiency would result in a decrease in net GHG balance between 4.8% and 6.0%. The affect of changes in initial biogenic carbon stocks is also large: a 10% increase in initial carbon stocks produces an increase in net GHG balance between 5.1% and 8%; similarly, a decrease in initial carbon stocks of 10% can lower the net GHG balance by 5.2% – 8.2%. This indicates that improvements in both biomass yield and conversion efficiency result in benefits to the GHG balance of low emissions projects. This also shows that the initial carbon stocks are an important contributor to the carbon balance, and as such should be always included in life cycle GHG analyses of biofuel projects. This can also mean that biofuel projects can improve their GHG balance by simply being sited in land areas low in C stocks, both live biomass and dead organic matter.

For ethanol production cost, the linear regression sensitivity analysis suggests that the variability in ethanol cost is most sensitive to the variability in conversion efficiency, and much less sensitive to the other input variables across their range of values. The variability in biomass yield contributes less than 4% to the variability in ethanol cost, and conversion yield less than 8%. A 10% improvement in ethanol conversion efficiency results in a 5.3-5.7% reduction in ethanol production cost, while a 10% improvement in biomass yield reduces ethanol cost by 2.1-2.2%.

For plantation area, the linear regression sensitivity analysis for the range of values of input variables suggests that the variability in area is most sensitive to biorefinery capacity, and less sensitive to biomass yield and conversion efficiency. The latter two account for less than 6% and 2% of the variability in area, respectively. This is explained by the large ranges variation for biorefinery capacity (±67%) compared with the smaller ranges for biomass yield (±18%) and conversion efficiency (±13%). However, one-factor-at-a-time changes in all the three input variables produce similar changes in plantation area. A 10% improvement in either biomass yield or conversion efficiency results in a 9.1% decrease in plantation area, and a 10% increase in biorefinery capacity results in a 10% increase in area.

In conclusion, this study offers important information for policy makers as well as for research funding agencies. If the goal of biofuel development policies is to penetrate the transportation fuels market through low production cost of ethanol, then the improvements in ethanol conversion efficiency will deliver greater cost reductions (nearly three-fold) than improvements in biomass yield.

If one of the goals of biofuel policies is to reduce the plantation area required, then the improvements in biomass yield will result in comparable benefits as the improvements in conversion efficiency.

If the aim of biofuel policies is the reduction of GHG emissions, however, the impact of improvements is different. Biofuel projects that have a high net GHG balance will indeed benefit from the improvements in biomass yield and conversion efficiency. However, if biofuel projects already have a low net GHG balance, the benefits from improvements will be small. In fact, improving the conversion efficiency may actually worsen the GHG balance for low emissions projects that draw benefits from carbon sequestration in live biomass and organic matter.

CHAPTER 6 Conclusions and recommendations for future work

6.1 Conclusions

This dissertation has accomplished all the five research objectives proposed in section 1.4. For the first objective, the Carbon-aware Biomass production Optimization System (C-BOS) was developed in CHAPTER 2. C-BOS is a biomass production planning model for fast growing tree plantations, which can be used for tactical analysis of biomass-to-biofuel systems. The C-BOS modeling framework allows the subsequent quantification and monitoring of organic matter stocks dynamics and greenhouse gas fluxes, in a time-dependent manner and from a life cycle perspective. Besides the harvesting/planting planning sequence of activities, the C-BOS model also determines the plantation area needed and the ethanol production cost.

To our knowledge C-BOS is the first implementation of a linear programming harvest scheduling model Type I for a life cycle analysis of biofuel production systems, where individual parcels of land are not aggregated with other land parcels over the planning horizon. We consider this to be a significant contribution to other current approaches that used a modeling Type II structure (Hennigar et al. 2008, Neilson et al. 2008). In model type II, when land parcels are being grouped together based on their harvest ages (and implicitly on the merchantable stem carbon pool), all other carbon pools are also grouped together. However, on non-homogenous landscapes (*i.e.* with different soil and DOM carbon stocks) it is unlikely that these carbon pools would have the same amounts of stored carbon across the land parcels being grouped, and therefore this information is "lost" for future iterations. This "loss of history" for the carbon pools stocks and flows makes it impracticable to accurately track carbon stocks and dynamics through time in a model type II structure. In contrast, the model type I used in the C-BOS model developed in this dissertation enables the subsequent monitoring of biomass (carbon) dynamics for biogenic carbon pools within each land parcel, both live biomass pools and DOM pools. In terms of live biomass, C-BOS monitors not only the harvested biomass, but all the biomass pools (stem, bark, branch, foliage, coarse roots, fine roots). The explicit monitoring of individual parcels and their respective biomass pools through the planning horizon allows for the determination of biogenic carbon sequestration during tree growth, as well as the biogenic carbon removal at harvest time. This is key for determining the biogenic carbon balance of the biofuel project by directly accounting for the carbon transfers between atmosphere and the organic matter produced by the biofuel project.

For the second objective, the life cycle Biogenic Carbon balance and Dynamics model (Bio-CarbD) was developed (CHAPTER 3). Bio-CarbD uses as input the planning information resulting from the C-BOS model, and simulates the life cycle biogenic carbon balance of a biofuel project, and the dynamics of biogenic carbon pools by individual land planning units throughout the planning horizon of the project. Bio-CarbD monitors all the carbon pools included in the CBM-CFS3 model, which is a contribution to other approaches which do not include the slow and very slow decaying DOM pools, and combined the remaining pools into a few amalgamated carbon pools (Hennigar *et al.* 2008, Neilson *et al.* 2008).

A novel approach of the Bio-CarbD model is that it does not make any assumptions about the "carbon-neutrality" of the biomass generated by the biofuel project. Instead, Bio-CarbD accounts for all the biogenic carbon absorption from the atmosphere through natural biomass growth processes, the carbon transfers between live biomass pools and dead organic matter pools, the carbon emissions from biodegradation of organic material through both DOM decay and biomass to biofuel conversion processes, as well as the carbon emissions from the end use of the ethanol at the end of the life cycle.

Another novel approach of the implementation of the "one inventory plus change" methodology in Bio-CarbD is that the carbon pools are defined and monitored within individual land parcels, which are kept intact throughout the planning horizon (*i.e.* hectares of one land planning unit are not broken up and combined with hectares from other planning units that are harvested at the same time). As mentioned earlier, other adaptations of the CBM-CFS3 method (Hennigar *et al.* 2008, Neilson *et al.* 2008) used a forest management planning model that is designed to allow for breaking up the hectares of land units and combining them with hectares from other land units that happen to be harvested in the same year. However, hectares that are broken up and combined may have very different sizes of carbon pools even if they are harvested at the same time. When these different size carbon pools are combined together the information about the actual carbon quantity in the resulting hectares is lost.

The eight scenarios modeled with Bio-CarbD in CHAPTER 3 indicate that all three categories of scenario inputs (improvements in biomass yield and conversion efficiency, idle treatments, and initial carbon stocks and soil carbon losses from mechanized activities) affect the life cycle carbon balance of biofuel projects: i) the improvements in biomass yield and conversion efficiency appear to not necessarily result in more carbon sequestration or less emissions, when the objective is minimizing project costs with no consideration of carbon emissions/sequestration implications; ii) when biofuel projects use production strategies that pursue project cost minimization and allow idle treatments (towards the end of the planning horizon) to achieve that primary objective, this may result in less carbon sequestered over the life cycle of the project, which is not a viable climate change mitigation strategy; iii) the initial carbon stocks and carbon losses or emissions due to land-use change were an important component of the carbon balance of biofuel projects; this outcome held in our analysis even in the case when the biomass yield and conversion efficiency improved over time, hence suggesting that life-cycle analyses can be improved if these factors are considered.

For the third objective, the Carbon Balance and Biomass to Biofuel Optimization planning model (C3BO) was developed (CHAPTER 4). C3BO determines the net greenhouse gas balance of biofuel projects on a life cycle basis (from the initial land-use change, establishment of plantations and construction of biorefinery, through the conversion of biomass into ethanol and the final use in internal combustion engines) throughout the project time horizon, including biogenic carbon as well as all other greenhouse gas emissions generated by the project, and any additional credits for substituting fossil fuels. By comparing the life-cycle GHG emissions balance with that of the displaced fossil fuel, C3BO is able to quantify the viability of the biofuel project as a climate change mitigation strategy.

A novel approach of the C3BO model framework is the use of a biogenic carbon balance method in the calculation of the project net GHG balance, in addition to the energy input-output approach commonly used in the literature. By combining the C-BOS and Bio-CarbD models in one framework, C3BO allows for the analysis of the impacts of different biomass and ethanol production strategies not only on the biogenic carbon dynamics, but also on the overall life-cycle project net GHG balance.

Other novel approaches of the C3BO model are:

- a project-level methodology that addresses the GHG impacts of a biofuel project compared with the absence of the project;
- the consideration of initial land-use change effects not only on the live biomass carbon pools, but also on DOM and soil carbon pools; modeling the dynamics of soil biogenic carbon sequestration and emissions for each land parcel area over time;
- modeling the potential affect of harvesting/planting mechanized activities on soil carbon emissions through soil compaction and scarification;
- modeling the sequestration of biogenic carbon not only in live biomass, but also in DOM pools; this can reveal situations when the biofuel project results in net losses (emissions) of biogenic carbon, but also in net gains (sequestration) in biogenic carbon, results which would be totally missed by simply assuming that biomass is "carbon neutral;"
- inclusion of both i) carbon removals from atmosphere through biomass growth and ii) biogenic carbon releases back into atmosphere through biomass removal/harvesting, eliminates the need to assume the disputed notion of whether biomass is carbon neutral or not.

For the fourth objective, the conditions when a biofuel production system can be a viable climate mitigation strategy have been discussed in CHAPTER 4. The model C3BO is able to identify the conditions when a biofuel project increases the terrestrial carbon stocks and reduces the greenhouse gas emissions relative to a baseline determined by the comparable fossil fuel production system displaced; conversely, C3BO is also able to identify the conditions when displacing gasoline with wood ethanol from short rotation plantations is not a viable mitigation strategy.

To utility of the C3BO model was illustrated for a test case using a prototype wood-to-ethanol project. The low-emissions and high-emissions scenarios produced a wide range of results. The life cycle net GHG balance of wood-to-ethanol projects can range from being largely negative (meaning that the ethanol project emissions are much lower than those of the displaced gasoline reference system), to being largely positive (the ethanol project emissions are much larger than what have happened in the absence of the project), or somewhere in between.

A comparison with other published studies showed that the C3BO test case results include the range of results from other studies, except for the very low values reported for cellulosic ethanol by Farrell *et al.* (2006). Results from the C3BO model show that the potential direct land-use change impacts of biofuel projects (including the initial biomass removal, but perhaps more importantly the affect of the project on the organic matter balance and emissions in the soil) can be an important component of the life-cycle GHG balance, therefore suggesting that they need to be included in GHG balance analyses of biofuel projects. C3BO results suggest that it can take several decades for the biofuel project to offset the initial emissions from land-use change, through carbon sequestration in biomass and soil, and displacement of fossil fuel emissions. Our results also indicate that the magnitude of the reductions in emissions, compared with the fossil fuel baseline, is highly dependent on the length of the time frame considered for the biofuel project; *i.e.* project emissions at 60 years can be substantially different than at 100 years.

Previous reports published in the literature about life cycle GHG emissions of ethanol projects found either that displacing gasoline with ethanol would result in less GHG emissions, or that it would result in more GHG emissions, than the absence of the ethanol project. On one side of the argument are studies who suggest that wood ethanol projects have a lower GHG emissions balance than the displaced gasoline, and the question is not “if” ethanol is better than gasoline but “by how much”. On the other side there are the studies that argue that ethanol may result in higher emissions than the displaced fossil fuel.

The results from the C3BO model suggest that both viewpoints are correct, as discussed in CHAPTER 4: biofuel projects can indeed produce fewer emissions than the fossil fuel baseline under defined conditions, but they can also result in more emissions under other specific conditions. In other words, the viability of biofuel projects as worthwhile climate mitigation strategies depends on project-specific conditions that need to be properly assessed on a project-by-project basis.

The importance of DOM decay emissions to the net GHG balance of the biofuel project was also analyzed (CHAPTER 4). The value of the DOM decay emissions was (in some cases much) larger in absolute terms than the net GHG balance for all scenarios analyzed. In two of the scenarios there was less biogenic carbon stored in pools at the end of the project compared with the beginning of the project. This was due to the large carbon removals from DOM pools through natural decomposition processes, as a direct result of the project activities.

The importance of including the emissions from DOM decay in the overall biogenic carbon balance was evident in two of the scenarios analyzed in CHAPTER 4, where the emissions from biomass removal and from DOM decay exceeded the carbon dioxide captured from the atmosphere. This means that the project not only did not sequester any net carbon in live or dead organic matter pools at the end of the time horizon, but actually released carbon dioxide into atmosphere that was stored in the DOM pools before land-use change.

If the biomass had been simply considered "carbon-neutral", then none of the emissions from DOM decay would have been accounted for. On one hand, the analysis would not have identified that the project actually resulted in net emissions in the two scenarios, from the biogenic carbon perspective. On the other hand, an assumption of biomass "carbon neutrality" would not have identified that in the other two scenarios the project actually resulted in a net sequestration of carbon in live and dead organic matter pools, which contributed to the project to being a viable mitigation strategy.

Finally, for the fifth objective, the impact and relative importance of C3BO input variables on the biofuel project net greenhouse gas balance for several project time horizons, as well as on plantation area and on ethanol production cost, were determined and discussed in CHAPTER 5. For the input variables ranges considered, the sensitivity analysis using linear regression suggests that the net GHG balance is most sensitive to ethanol production emissions and initial carbon stocks, and much less sensitive to the other input variables. As the project time horizon increases, the impact of initial carbon stocks decreases, and that of ethanol production emissions increases. Setting the biofuel project in land areas that have a low carbon stock can result in benefits that are higher than those obtained by improvements in biomass yield and conversion efficiency. Even for long time horizons such as 100 years, a 10% reduction in the initial C stock can result in a 5.2% improvement in the GHG balance. This shows the importance of including the initial carbon stocks in the life cycle GHG analyses of biofuels.

The improvements in biomass yield and conversion efficiency both contribute to reductions in ethanol production cost. However, these improvements do not necessarily result in improved GHG balances. When a biofuel project is set in conditions that generate a very high GHG balance, the impact of changes in biomass yield and ethanol conversion efficiency is large: a 10% improvement in biomass yield and conversion efficiency will contribute to reductions in the GHG balance in the range of 4.3-6.7% and 4.8-6.0% respectively.

However, when the net GHG balance of a biofuel project is very low, changes in biomass yield and initial carbon stocks have a small impact on the GHG balance. Improvements in conversion efficiency can actually worsen the GHG balance of biofuel projects, which benefit from carbon accumulation on land, in live biomass and dead organic matter. A 10% increase in conversion efficiency can worsen the GHG balance by 4.1-7.6%.

A key conclusion of this study is that displacing gasoline with wood ethanol from fast growing tree plantations may not always be a viable climate mitigation strategy; however, many conditions seem to exist where the fuel switch can indeed be viable. We suggest that the question of the viability of the fuel switch does not have a generic predisposition “wood ethanol always has lower GHG emissions than gasoline” or “always higher” GHG emissions than gasoline. In short, it depends on project-specific input parameters, and therefore the viability must be determined on a project-by-project basis.

6.2 Potential applications and recommendations for future research

The results of this research can be a source of essential information for developing policies for large-scale wood energy production systems, as well as for determining the viability of project-specific conditions for displacing gasoline with wood ethanol.

The C3BO modeling framework proposed in this dissertation can enable project proponents and policy makers to evaluate different scenarios to understand trade-offs between life-cycle costs and benefits, and to evaluate opportunities to maximize the project's financial utility while reducing negative land base impacts, reducing carbon dioxide emissions, and increasing biomass and land base carbon stocks throughout the planning horizon.

The three modeling frameworks described herein permit expanded analyses and potential applications beyond the limited scope of the test cases that were presented here.

First, the C-BOS model formulation allows for additional biomass production strategies to be analyzed: more land unit types could be modelled than the six types considered in the test case; the differences in land productivity could be modelled as decision variables; the land area available could be constrained for each type of land unit; the analysis can be extended for project time horizons longer than 68 years; more treatment types (biomass production strategies) could be modelled than the six types considered. Since the objective function of the optimization model is to minimize production costs, a potential further area of research could be to include the minimization of project net GHG emissions and the minimization of plantation area, in a multi-objective type of formulation, which would aim to reduce emissions and land use at the same as it reduces costs.

Second, the structure of the Bio-CarbD model allows for expanded applications such as: analyzing project time horizons longer than 68 years; modeling more than one land productivity type; focusing the analysis either only on irrigated or on non-irrigated land unit types, instead of mixing the two; instead of analyzing the impact of changing input variables at the same time, one could consider separately the impacts of 1) initial carbon stocks, 2) carbon losses at land-use change, and 3) carbon losses from mechanized activities, on the project life cycle carbon balance. Similarly for the yield improvements, the separate effect of biomass yield and of conversion efficiency could be investigated.

Also, if the carbon-GHG balance of the biofuel project needs to be calculated for all plantation areas, and for the entire duration of the project horizon, then the question arises how to model the carbon in the land parcels that have the idle treatment applied to them; *i.e.* what happens with the biogenic carbon in the respective land parcels since no more biomass is purposely grown on those land parcels after the idle treatment is applied. This aspect, not analyzed in this dissertation, is a potential further area of research.

Another potential area of research is to study the value of CBM-CFS3 parameters (e.g., live biomass turnover and litterfall parameters, DOM decay rates and physical transfer parameters) to be used for the particular conditions of short rotation plantations, since these parameters have not been specifically selected or researched for these situations within CBM-CFS3.

Third, the framework of the C3BO model can be used for more complex analyses by turning on certain features or inputs that were somewhat restricted for the purposes of the research in CHAPTER 4 and CHAPTER 5. For example, the impact of making the idle treatments available at the end of the planning horizon can be studied; the sensitivity analysis can be expanded to include the non-irrigated coppice, and the irrigated single stem and coppice treatments.

Bibliography

(S&T)2 Consultants Inc. 2011. GHGenius – a model for lifecycle assessment of transportation fuels, version 3.19. Natural Resources Canada. Ottawa, ON.

(S&T)2 Consultants Inc. 2010. GHGenius – a model for lifecycle assessment of transportation fuels, version 3.17. Natural Resources Canada. Ottawa, ON.

(S&T)2 Consultants Inc. 2010. Analysis of biofuel pathways using GHGenius 2010. Natural Resources Canada, Office of Energy Efficiency. Ottawa, ON.

(S&T)2 Consultants Inc. 2003. Documentation for Natural Resources Canada's GHGenius Model. Natural Resources Canada. Ottawa, ON.

Aden, A., Ruth, M., Ibsen, K., Jechura, J., Neeves, K., Sheehan, J., Wallace, B., Montague, L., Slayton, A., and Lukas, J. 2002. Lignocellulosic Biomass to Ethanol Process Design and Economics Utilizing Co-Current Dilute Acid Prehydrolysis and Enzymatic Hydrolysis for Corn Stover. NREL - National Renewable Energy Laboratory. Rep. NREL/TP-510-32438.Golden, CO.

Adler, P.R., S.J. Del Grosso, and W.J. Parton. 2007. Life-cycle assessment of net greenhouse-gas flux for bioenergy cropping systems. *Ecol. Appl.* 17: 675-691.

Afas, N.A., N. Marron, S. Van Dongen, et al. 2008. Dynamics of biomass production in a poplar coppice culture over three rotations (11 years). *For. Ecol. Manage.* 255(5-6): 1883-1891.

Agostini, A., Giuntoli, J., and Boulamanti, A. 2013. Carbon accounting of forest bioenergy - Conclusions and recommendations from a literature review. European Commission, Joint Research Centre, Institute for Energy and Transport.

Akaike, H. 1974. A new look at the statistical model identification. *Automatic Control, IEEE Transactions on* 19(6): 716-723.

Alper, H., J. Moxley, E. Nevoigt, et al. 2006. Engineering Yeast Transcription Machinery for Improved Ethanol Tolerance and Production. *Science* 314(5805): 1565-1568.

Ammann, C., C.R. Flechard, J. Leifeld, et al. 2007. The carbon budget of newly established temperate grassland depends on management intensity. *Agriculture, Ecosystems & Environment* 121: 5-20.

Apps, M.J., W.A. Kurz, S.J. Beukema, et al. 1999. Carbon budget of the Canadian forest product sector. *Environ. Sci. & Policy* 2(1): 25-41.

Audsley, E., and J.E. Annetts. 2003. Modelling the value of a rural biorefinery—part I: the model description. *Agricultural Systems* 76(1): 39-59.

Aylott, M.J., E. Casella, I. Tubby, et al. 2008. Yield and spatial supply of bioenergy poplar and willow short-rotation coppice in the UK. *New Phytol.* 178(2): 358-370.

BC Assessment. 2007. Farm Classification in British Columbia. BC Assessment, Provincial Crown Corporation. Available at: http://www.bcassessment.ca/public/Fact%20Sheets/Farm%20Classification%20in%20British%20Columbia.pdf. Retrieved on: December 2012.

BC Assessment. Classifying Farm Land. Available at: http://bcassessment.gov.bc.ca/process/agricultural_forestry/classify_farm.asp. Accessed: 2011 January.

BC MoE. 2011. Methodology for Reporting B.C. Public Sector Greenhouse Gas Emissions. Version 1.0. B.C. Ministry of Environment. Victoria, BC.

BC MoEMPR. 2008. BC Bioenergy Strategy: Growing our natural energy advantage. BC Ministry of Energy, Mines and Petroleum Resources. Victoria, B.C.

BC MoEMPR. 2008. The BC Energy Plan: A vision for clean energy leadership. BC Ministry of Energy, Mines and Petroleum Resources. Victoria, B.C. Available at: http://www.energyplan.gov.bc.ca/PDF/BC_Energy_Plan.pdf. Retrieved on: August 2012.

BC Stats. 1999. Business Indicators February 1999. BC Stats. Rep. 99-02.Victoria, B.C. Available at: http://www.bcstats.gov.bc.ca/pubs/bcbi/bcbi9902.pdf. Retrieved on: June 2010.

Bill Berguson. 2010. Personal Communication. Natural Resources Research Institute, University of Minnesota, Duluth, MN 55811, USA.

Berthrong, S.T., E.G. Jobbagy, and R.B. Jackson. 2009. A global meta-analysis of soil exchangeable cations, pH, carbon, and nitrogen with afforestation. *Ecological Applications* 19: 2228-2241.

Bessou, C., B. Mary, J. Leonard, et al. 2010. Modelling soil compaction impacts on nitrous oxide emissions in arable fields. *European Journal of Soil Science* 61(3): 348-363.

BIOCAP Canada. 2008. An information guide on pursuing biomass energy opportunities and technologies in British Columbia. BC Ministry of Energy, Mines and Petroleum Resources.

BIOCAP Canada. 2006. An Inventory of the Bioenergy Potential of British Columbia. BIOCAP Canada. Kingston, ON.

Birdsey, R.A. 1992. Carbon storage and accumulation in United States forest ecosystems. USDA Forest Service. Rep. General Technical Report WO-59.Washington, D.C.

Birdsey, R.A. 2006. Carbon Accounting Rules and Guidelines for the United States Forest Sector. *J. Environ. Qual*. 35(4): 1518-1524.

Black, T.A., and J.W. Harden. 1995. Effect of timber harvest on soil carbon storage at Blodgett Experimental Forest, California. *Can. J. For. Res*. 25: 1385-1396.

Bonin, C., and R. Lal. 2012. Bioethanol Potentials and Life-Cycle Assessments of Biofuel Feedstocks. *Critical Reviews in Plant Sciences* 31(4): 271-289.

Borrion, A.L., M.C. McManus, and G.P. Hammond. 2012. Environmental life cycle assessment of lignocellulosic conversion to ethanol: A review. *Renewable and Sustainable Energy Reviews* 16(7): 4638-4650.

Brandao, M., A. Levasseur, M.F. Kirschbaum, et al. 2013. Key issues and options in accounting for carbon sequestration and temporary storage in life cycle assessment and carbon footprinting. *The International Journal of Life Cycle Assessment* 18(1): 230-240.

Brandão, M., L. Milà i Canals, and R. Clift. 2011. Soil organic carbon changes in the cultivation of energy crops: Implications for GHG balances and soil quality for use in LCA. *Biomass Bioenergy* 35(6): 2323-2336.

Broekoff, D., and Zyla, K. 2008. Outside the cap: opportunities and limitations of greenhouse gas offsets. World Resources Institute. Washington, DC.

Buchholz, T., and T. Volk. 2011. Improving the Profitability of Willow Crops—Identifying Opportunities with a Crop Budget Model. *BioEnergy Research* 4(2): 85-95.

Bull, G.Q., M. Bazett, O. Schwab, et al. 2006. Industrial forest plantation subsidies: Impacts and implications. *Forest Policy and Economics* 9(1): 13-31.

California Environmental Protection Agency. 2009. Detailed California-Modified GREET Pathway for Cellulosic Ethanol from Farmed Trees by Fermentation (DRAFT FOR REVIEW). California Environmental Protection Agency, Air Resources Board, Stationary Source Division. Rep. Version 2.1.

Campbell, J.E., D.B. Lobell, and C.B. Field. 2009. Greater Transportation Energy and GHG Offsets from Bioelectricity Than Ethanol. *Science* 324(5930): 1055-1057.

Canadian Renewable Fuels Association. 2010. Growing Beyond Oil Delivering Our Energy Future - A Report Card on the Canadian Renewables Fuels Industry. Canadian Renewable Fuels Association. Ottawa, ON. Available at: http://www.greenfuels.org/uploads/documents/crfareportcardenglish2010final.pdf. Retrieved on: January 2013.

Canals, L., C. Bauer, J. Depestele, et al. 2007. Key Elements in a Framework for Land Use Impact Assessment Within LCA. *The International Journal of Life Cycle Assessment* 12(1): 5-15.

Michael Carlson. 2011. Personal Communication. BC Ministry of Forests and Range.

Dan Carson. 2009. Personal Communication. Kruger Products Ltd.

Caterpillar Inc. 2011. Caterpillar Performance Handbook. Caterpillar Inc. Available at: http://www.albancat.com/UserFiles/Uploaded/cms/cat-performance-handbook-43.pdf. Retrieved on: May 2012.

Cathcart, J., and Delaney, M. 2006. Carbon Accounting: Determining carbon offsets from forest projects. The Oregon Forest Resources Institute. Portland, OR.

Chambers, R.S., R.A. Herendeen, J.J. Joyce, et al. 1979. Gasohol: Does It or Doesn't It Produce Positive Net Energy? *Science* 206(4420): 789-795.

Chaudhuri, S.K., and D.R. Lovley. 2003. Electricity generation by direct oxidation of glucose in mediatorless microbial fuel cells. *Nature Biotechnology* 21: 1229-1232.

Cherubini, F., G.P. Peters, T. Berntsen, et al. 2011. CO2 emissions from biomass combustion for bioenergy: atmospheric decay and contribution to global warming. *GCB Bioenergy* 3(5): 413-426.

Cherubini, F. 2010. GHG balances of bioenergy systems – Overview of key steps in the production chain and methodological concerns. *Renewable Energy* 35(7): 1565-1573.

Cherubini, F., N.D. Bird, A. Cowie, et al. 2009. Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations. *Resour. Conserv. Recycling* 53(8): 434-447.

Climate Action Reserve. 2010. Forest Project Protocol Version 3.2.

Climate Action Secretariat, Gov't. of B.C. 2008. Climate Action Plan. Government of British Columbia, Canada. Victoria, B.C. Available at: http://www.livesmartbc.ca/attachments/climateaction_plan_web.pdf. Retrieved on: April 2013.

Coleman, H.D., J. Park, R. Nair, et al. 2008. RNAi-mediated suppression of p-coumaroyl-CoA 3′-hydroxylase in hybrid poplar impacts lignin deposition and soluble secondary metabolism. *Proceedings of the National Academy of Sciences* 105(11): 4501-4506.

Coleman, H.D., T. Canam, K. Kang, et al. 2007. Over-expression of UDP-glucose pyrophosphorylase in hybrid poplar affects carbon allocation. *J. Exp. Bot.* 58(15-16): 4257-4268.

Covington, W.W. 1981. Changes in the forest floor organic matter and nutrient content following clear cutting in northern hardwoods. *Ecology* 62(1): 41-48.

Cowie, A., P. Smith, and D. Johnson. 2006. Does Soil Carbon Loss in Biomass Production Systems Negate the Greenhouse Benefits of Bioenergy? *Mitigation Adapt. Strat. Global Change* 11(5): 979-1002.

Crawley, M.J. 2005. Statistics: An Introduction Using R. J. Wiley,

Crutzen, P.J., A.R. Mosier, K.A. Smith, et al. 2007. N2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. *Atmospheric Chemistry and Physics Discussions* 7(4): 11191-11205.

Czitrom, V. 1999. One-Factor-at-a-Time versus Designed Experiments. *The American Statistician* 53(2): 126-131.

Dale, B.E. 2008. Biofuels: Thinking Clearly about the Issues. *J. Agric. Food Chem.* 56(11): 3885-3891.

Dale, B.E. 2007. Thinking clearly about biofuels: ending the irrelevant "net energy" debate and developing better performance metrics for alternative fuels. *Biofuels, Bioproducts and Biorefining* 1(1): 14-17.

Dale, V.H., Kline, K.L., Wiens, J., and Fargione, J. 2010. Biofuels: Implications for Land Use and Biodiversity. Ecological Society of America. Washington, D.C. Available at: http://www.esa.org/biofuelsreports. Retrieved on: June 2012.

Danzco Inc. Pull Through Delimbers. Available at: http://www.danzcoinc.com/html/pull_though_delimbers.html. Accessed: 2012 February.

Davis, L.S., Johnson, K.N., Bettinger, P., and Howard, T.E. 2001. Forest Management to Sustain Ecological, Economic, and Social Values. Waveland Press, Inc., Long Grove, IL.

Deckmyn, G., I. Laureysens, J. Garcia, et al. 2004. Poplar growth and yield in short rotation coppice: model simulations using the process model SECRETS. *Biomass and Bioenergy* 26(3): 221-227.

Deere & Company. 4630 Self-Propelled Sprayer. Available at: http://www.deere.com/wps/dcom/en_US/products/equipment/self_propelled_sprayers/4630/4630.page. Accessed: 2012 March.

Dehue, B., Meyer, S., and Hamelinck, C. 2007. Towards a harmonised sustainable biomass certification scheme. Ecofys Netherlands BV. Utrecht, the Netherlands.

Delucchi, M.A. 2003. A Lifecycle Emissions Model (LEM): Lifecycle Emissions from Transportation Fuels, Motor Vehicles, Transportation Modes, Electricity Use, Heating and Cooking Fuels, and Materials. Rep. UCD-ITS-RR-03-17.

Delucchi, M. 2011. A conceptual framework for estimating the climate impacts of land-use change due to energy crop programs. *Biomass Bioenergy* 35(6): 2337-2360.

Demirbas, A. 2003. Sustainable cofiring of biomass with coal. *Energy Conversion and Management* 44(9): 1465-1479.

Dickmann, D.I. 2006. Silviculture and biology of short-rotation woody crops in temperate regions: Then and now. *Biomass Bioenergy* 30(8-9): 696-705.

DiPardo, J. 2004. Outlook for Biomass Ethanol Production and Demand. U.S. Energy Information Administration. Available at: http://www.agmrc.org/media/cms/biomass_E6EE9065FD69D.pdf. Retrieved on: May 2009.

Djomo, S.N., O.E. Kasmioui, and R. Ceulemans. 2011. Energy and greenhouse gas balance of bioenergy production from poplar and willow: a review. *CGB Bioenergy* 3: 181-197.

Douglas, C., and Mansfield, S.D. 2009. GENOME BC project: Optimized populus feedstocks and novel enzyme systems for a British Columbian bioenergy sector. Genome British Columbia. Available at: http://www.genomebc.ca/portfolio/projects/bioenergy-projects/completed/optimized-populus-feedstocks-and-novel-enzyme-systems-for-a-brit/. Retrieved on: December 2013.

Duffy, G., and Beale, N. 2005. Energy from Biomass: Summaries of Biomass Projects carried out as part of the DTI's Technology Programme: New and Renewable Energy. Department of Trade and Industry (United Kingdom). Rep. URN NUMBER: 05/1110.

Jake Eaton. 2010. Personal Communication. GreenWood Resources Inc.

Edwards, W. 2009. Estimating Farm Machinery Costs. Iowa State University, University Extension. Rep. PM-710.

Eggeman, T., and R.T. Elander. 2005. Process and economic analysis of pretreatment technologies. *Bioresour. Technol*. 96(18): 2019-2025.

Environment Canada. 2010. National Inventory Report 1990–2008: Greenhouse Gas Sources and Sinks in Canada. Available at: http://www.ec.gc.ca/ges-ghg/default.asp?lang=En&n=1357A041-1. Retrieved on: September 2013.

Environment Canada. Renewable Fuels - Ethanol. Available at: http://www.ec.gc.ca/energie-energy/default.asp?lang=En&n=BDB8F633-1. Last updated: 2010. Accessed: 2012 March.

Environment Canada. Emission factors for industrial processes from Canada's National Inventory report. Available at: http://www.ec.gc.ca/ges-ghg/default.asp?lang=En&n=1A18593B-1. Accessed: 2011 March.

European Commission. 2011. First EU sustainability schemes for biofuels get the go-ahead. Rep. IP/11/901.Available at: http://europa.eu/rapid/pressReleasesAction.do?reference=IP/11/901&format=HTML&aged=0&language=EN&guiLanguage=en;. Retrieved on: June 2012.

European Council. 2009. Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources. Official Journal of the European Union. Rep. Directive 2009/28/EC.Available at: http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2009:140:0016:0062:EN:PDF. Retrieved on: August 2013.

Fang, S., X. Xu, S. Lu, et al. 1999. Growth dynamics and biomass production in short-rotation poplar plantations: 6-year results for three clones at four spacings. *Biomass Bioenergy* 17(5): 415-425.

FAO. 1992. Cost Control in Forest Harvesting and Road Construction. Food and Agriculture Organization of the United Nations. Available at: http://www.fao.org/docrep/T0579E/t0579e00.htm#Contents. Retrieved on: February 2012.

Fargione, J., J. Hill, D. Tilman, et al. 2008. Land Clearing and the Biofuel Carbon Debt. *Science* 319(5867): 1235-1238.

Farrell, A.E., R.J. Plevin, B.T. Turner, et al. 2006. Ethanol Can Contribute to Energy and Environmental Goals. *Science* 311(5760): 506-508.

Fearnside, P.M. 2002. Why a 100-year time horizon should be used for global warming mitigation calculations. *Mitig Adapt Strateg Glob Chang* 7(1): 19-30.

Feldman, B. 2005. Relative Importance and Value. Rep. Manuscript version 1.1, 2005-03-19.

Fernandes, E.C.M., P.P. Motavalli, C. Castilla, et al. 1997. Management control of soil organic matter dynamics in tropical land-use systems. *Geoderma* 79(1–4): 49-67.

Field, C.B., J.E. Campbell, and D.B. Lobell. 2008. Biomass energy: the scale of the potential resource. *Trends in Ecology & Evolution* 23(2): 65-72.

Field, F., R. Kirchain, and J. Clark. 2001. Life-cycle assessment and temporal distributions of emissions. *J. Ind. Ecol.* 4(2): 71-91.

Forest Carbon Standards Committee. 2009. Draft Forest Carbon Standard (Update: November 9, 2009).

Forsberg, G. 2000. Biomass energy transport: Analysis of bioenergy transport chains using life cycle inventory method. *Biomass Bioenergy* 19(1): 17-30.

Fredeen, A.L., C.H. Bois, D.T. Janzen, et al. 2005. Comparison of coniferous forest carbon stocks between old-growth and young second-growth forests on two soil types in central British Columbia, Canada. *Canadian Journal of Forest Research* 6

Frey, H.C., and S.R. Patil. 2002. Identification and Review of Sensitivity Analysis Methods. *Risk Analysis: An International Journal* 22(3): 553-578.

Frieden, D., N. Pena, and D.N. Bird. 2012. Incentives for the use of forest biomass: a comparative analysis of Kyoto Protocol accounting pre- and post-2012. *Greenhouse Gas Measurement and Management* 2(2-3): 84-92.

Frischknecht, R., Jungbluth, N., Althaus, H.-., Doka, G., Dones, R., Hischier, R., Hellweg, S., Nemecek, T., Rebitzer, G., Spielmann, M., and Wernet, G. 2007. Overview and Methodology. Final report ecoinvent data v2.0, No. 1. Swiss Centre for Life Cycle Inventories. Dübendorf, CH. Available at: http://www.ecoinvent.org/fileadmin/documents/en/01_OverviewAndMethodology.pdf. Retrieved on: February 2013.

Frischknecht, R., N. Jungbluth, H.-. Althaus, et al. 2005. The ecoinvent Database: Overview and Methodological Framework. *Int J Life Cycle Assess* 10(1): 3-9.

Fritsche, U.R., Hunecke, K., Hermann, A., Schulze, F., and Wiegmann, K. 2006. Sustainable Standards for Bioenergy. WWF Germany. Frankfurt am Main.

Fu, G.Z., A.W. Chan, and D.E. Minns. 2003. Life Cycle assessment of bio-ethanol derived from cellulose. *The International Journal of Life Cycle Assessment* 8(3): 137-141.

Gautam, S., R. Pulkki, C. Shahi, et al. 2010. Economic and energy efficiency of salvaging biomass from wildfire burnt areas for bioenergy production in northwestern Ontario: A case study. *Biomass Bioenergy* 34(11): 1562-1572.

GENOME BC. 2007. Towards a bioenergy sector strategy. Genome British Columbia. Rep. 1.0.Vancouver, BC. Available at: http://www.genomebc.ca/files/2813/1171/9701/Towards_a_Bioenergy_Sector_Strategy.pdf. Retrieved on: January 2013.

Gershenson, A., Barsimantov, J., and EcoShift Consulting LLC. 2009. Accounting for Carbon in Soils. Climate Action Reserve. Available at: http://www.climateactionreserve.org/wp-content/uploads/2010/12/Accounting_for_Carbon_in_Soils-Forest_White_Paper.pdf;. Retrieved on: June 2012.

Gielen, D.J., M.A.P.C. de Feber, A.J.M. Bos, et al. 2001. Biomass for energy or materials?: A Western European systems engineering perspective. *Energy Policy* 29(4): 291-302.

GLOBE Foundation. 2007. The endless energy project: A blueprint for complete energy self-sufficiency in British Columbia. GLOBE Foundation. Vancouver, B.C. Available at: http://globe.ca/wp-content/uploads/2012/10/globe_endlessenreport.pdf. Retrieved on: June 2012.

Gnansounou, E., A. Dauriat, J. Villegas, et al. 2009. Life cycle assessment of biofuels: Energy and greenhouse gas balances. *Bioresour. Technol.* 100(21): 4919-4930.

Grant, T., Beer, T., Campbell, P.K., and Batten, D. 2008. Life Cycle Assessment of Environmental Outcomes and Greenhouse Gas Emissions from Biofuels Production in Western Australia. CSIRO and The Department of Agriculture and Food, Government of Western Australia. Rep. KN29A/WA/F2.9.

Greig, M., and Bull, G.Q. 2009. Carbon management in British Columbia's forests: opportunities and challenges. FORREX - Forum for Research and Extension in Natural Resources Society. Rep. Forrex series, 1495-9658.Kamloops, B.C.

Grigal, D.F., and W.E. Berguson. 1998. Soil carbon changes associated with short-rotation systems. *Biomass Bioenergy* 14(4): 371-377.

Grömping, U. 2006. Relative Importance for Linear Regression in R: The Package relaimpo. *Journal of Statistical Software* 17(1): 1-27.

Groode, T., and Heywood, J. 2006. Review of Corn Based Ethanol Energy Use and Greenhouse Gas Emissions. Rep. LFEE 2007-01 RP.

Gunn, E. 2007. Models for Strategic Forest Management. *In* Handbook Of Operations Research In Natural Resources. Springer US, pp. 317-341.

Gutierrez, A.P., and L. Ponti. 2009. Bioeconomic Sustainability of Cellulosic Biofuel Production on Marginal Lands. *Bulletin of Science, Technology & Society* 29(3): 213-225.

Guy, R.D., and Benowicz, A. 1998. Can Afforestation Contribute to a Reduction in Canada's Net CO2 Emissions? Report prepared for the Canadian Pulp and Paper Association.

Hahn-Hägerdal, B., M. Galbe, M.F. Gorwa-Grauslund, et al. 2006. Bio-ethanol – the fuel of tomorrow from the residues of today. *Trends Biotechnol.* 24(12): 549-556.

Hall, J.P. 2002. Sustainable production of forest biomass for energy. *The Forestry Chronicle* 78(3): 391-396.

Hall, R.B., M.E. Ostry, and N.E. Nordh. 1992. IEA joint trials: New lessons from old plantations. *Biomass Bioenergy* 2(1-6): 85-94.

Hamburg, S.P. 2000. Simple rules for measuring changes in ecosystem carbon in forestry-offset projects. *Mitigation Adapt. Strat. Global Change* 5(1): 25-37.

Hamelinck, C.N., G.v. Hooijdonk, and A.P. Faaij. 2005. Ethanol from lignocellulosic biomass: techno-economic performance in short-, middle- and long-term. *Biomass Bioenergy* 28(4): 384-410.

Hammerschlag, R. 2006. Ethanol's Energy Return on Investment: A Survey of the Literature 1990–Present. *Environ. Sci. Technol.* 40(6): 1744-1750.

Hansen, E.A. 1994. A guide for determining when to fertilize hybrid poplar plantations. U.S. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station. Rep. Research Paper NC-319.St. Paul, MN.

Hansen, E.A., and Netzer, D.A. 1985. Weed control using herbicides in short-rotation intensively cultured poplar plantations. U.S. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station. Rep. Research Paper NC-260.St. Paul, MN.

Hansen, E.A., Netzer, D.A., and Tolsted, D.N. 1993. Guidelines for Establishing poplar Plantations in the North-Central U.S. USDA Forest Service, North Central Forest Experiment Station. Rep. Research Note NC-363.

Hansen, E.A. 1993. Soil carbon sequestration beneath hybrid poplar plantations in the North Central United States. *Biomass Bioenergy* 5(6): 431-436.

Hartsough, B., R. Spinelli, S. Pottle, et al. 2000. Fiber Recovery with Chain Flail Delimbing/Debarking and Chipping of Hybrid Poplar. *International Journal of Forest Engineering* 11(2)

Heiman, M.K., and B.D. Solomon. 2007. Fueling U.S. Transportation: The Hydrogen Economy and Its Alternatives. *Environment* 49(8): 10-25.

Heller, M.C., G.A. Keoleian, and T.A. Volk. 2003. Life cycle assessment of a willow bioenergy cropping system. *Biomass and Bioenergy* 25(2): 147-165.

Heller, M.C., G.A. Keoleian, M.K. Mann, et al. 2004. Life cycle energy and environmental benefits of generating electricity from willow biomass. *Renewable Energy* 29(7): 1023-1042.

Hendrickson, C., Lave, L., and Matthews, H.S. 2006. Environmental Life Cycle Assessment of Goods and Services: An Input-Output Approach. Resources for the Future, Washington, DC.

Hendrickson, C., A. Horvath, S. Joshi, et al. 1998. Economic input-output models for environmental life-cycle assessment. *Env. Sci. & Tech.* 32(7): 184A-191A.

Hennigar, C.R., D.A. MacLean, and L.J. Amos-Binks. 2008. A novel approach to optimize management strategies for carbon stored in both forests and wood products. *For. Ecol. Manage.* 256(4): 786-797.

Hill, J. 2007. Environmental costs and benefits of transportation biofuel production from food- and lignocellulose-based energy crops. A review. *Agronomy for Sustainable Development* 27(1): 1-12.

Hill, J., E. Nelson, D. Tilman, et al. 2006. Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. *Proc. Nat. Acad. Sci. USA* 103(30)

Hill, J., S. Polasky, E. Nelson, et al. 2009. Climate change and health costs of air emissions from biofuels and gasoline. *Proc. Nat. Acad. Sci. USA* 106(6): 2077-2082.

Himmel, M.E., S. Ding, D.K. Johnson, et al. 2007. Biomass Recalcitrance: Engineering Plants and Enzymes for Biofuels Production. *Science* 315(5813): 804-807.

Huang, H., S. Ramaswamy, W. Al-Dajani, et al. 2009. Effect of biomass species and plant size on cellulosic ethanol: A comparative process and economic analysis. *Biomass Bioenergy* 33(2): 234-246.

Huber, G.W., S. Iborra, and A. Corma. 2006. Synthesis of Transportation Fuels from Biomass: Chemistry, Catalysts, and Engineering. *Chem. Rev.* 106(9): 4044-4098.

Huntington, T.G., and D.F. Ryan. 1990. Whole-tree-harvesting effects on soil nitrogen and carbon. *For. Ecol. Manage*. 31: 193-204.

ICF International. 2009. Lifecycle Greenhouse Gas Emissions due to Increased Biofuel Production. Methods and Approaches to Account for Lifecycle Greenhouse Gas Emissions from Biofuels Production Over Time. Peer Review Report July 31, 2009. Available at: http://epa.gov/otaq/renewablefuels/rfs2-peer-review-emissions.pdf. Retrieved on: August 2012.

IEA. 2010. World Energy Outlook 2010. Organisation for Economic Co-operation and Development,

IEA. 2010. Sustainable Production of Second-Generation Biofuels - Potential and perspectives in major economies and developing countries. OECD/IEA.

IEA. 2009. IEA Bioenergy Annual Report 2008. International Energy Agency.

IEA Bioenergy Executive Committee. 2002. Sustainable Production of Woody Biomass for Energy. Rep. IEA - 157.

IEA Bioenergy Task 38. 2009. Workshop Report "Land Use Changes due to Bioenergy - Quantifying and Managing Climate Change and other Environmental Impacts", 30 March – 01 April 2009. IEA Bioenergy, Task 38. Suomenlinna, Helsinki, Finland.

Ingerson, A., and Loya, W. 2008.
Measuring Forest Carbon: Strengths and Weaknesses of Available Tools. The Wilderness Society. Washington, D.C.

IPCC. 2007. Climate Change 2007: Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Synthesis Report. Cambridge University Press. New York, USA.

IPCC. 2007. Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. Cambridge, UK.

IPCC. 2007. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. Cambridge, UK, and New York, NY. http://ipcc.ch/publications_and_data/ar4/wg1/en/ch2s2-10.html. Retrieved on: July 2012.

IPCC. 2006. 2006 Intergovernmental Panel on Climate Change Guidelines for National Greenhouse Gas Inventories, Intergovernmental Panel on Climate Change. IGES. Japan.

Isebrands, J.G. 2007. Best Management Practices Poplar Manual For Agroforestry Applications in Minnesota. Environmental Forestry Consultants, LLC. New London, WI.

ISO. 2006. ISO 14040:2006 Environmental Management - Life Cycle Assessment - Principles and Framework. International Organization for Standardization,

ISO. 2006. ISO 14044:2006 Environmental Management - Life Cycle Assessment - Requirements and Guidelines. International Organization for Standardization.

Jamnick, M.S., and K.R. Walters. 1993. Spatial and temporal allocation of stratum-based harvest schedules. *Canadian Journal of Forest Research* 23(3): 402-413.

Jobbagi, E.G., and R.B. Jackson. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications* 10(2): 423-436.

Johnson, D.W., and P.S. Curtis. 2001. Effects of forest management on soil C and N storage: meta analysis. *Forest Ecology and Management* 140(2-3): 227-238.

Johnson, E. 2009. Goodbye to carbon neutral: Getting biomass footprints right. *Environ. Impact Assess. Rev.* 29(3): 165-168.

Johnson, K.N., and H.L. Scheurman. 1977. Techniques for prescribing optimal timber harvest and investment under different objectives - discussion and synthesis. Society of American Foresters,

Joshi, C.P., and S.D. Mansfield. 2007. The cellulose paradox — simple molecule, complex biosynthesis. *Curr. Opin. Plant Biol.* 10(3): 220-226.

Joslin, J.D., and S.H. Schoenholtz. 1997. Measuring the environmental effects of converting cropland to short-rotation woody crops: A research approach. *Biomass and Bioenergy* 13(4-5): 301-311.

Joss, B., R. Hall, D. Sidders, et al. 2008. Fuzzy-logic modeling of land suitability for hybrid poplar across the Prairie Provinces of Canada. *Environ. Monit. Assess.* 141(1): 79-96.

Jungbluth, N., Chudacoff, M., Dauriat, A., Dinkel, F., Doka, G., Faist Emmenegger, M., Gnansounou, E., Kljun, N., Schleiss, K., Spielmann, M., Stettler, C., and Sutter, J. 2007. Life cycle inventories of bioenergy. ecoinvent report No. 17. Swiss Centre for Life Cycle Inventories. Dübendorf, CH. http://db.ecoinvent.org/ecoquery/files/17_Bioenergy.pdf. Retrieved on: September 2012.

Kaul, M., G. Mohren, and V. Dadhwal. 2010. Carbon storage versus fossil fuel substitution: a climate change mitigation option for two different land use categories based on short and long rotation forestry in India. *Mitigation and Adaptation Strategies for Global Change* 15(4): 395-409.

Kauter, D., I. Lewandowski, and W. Claupein. 2003. Quantity and quality of harvestable biomass from Populus short rotation coppice for solid fuel use—a review of the physiological basis and management influences. *Biomass Bioenergy* 24(6): 411-427.

Kelliher, F.M., Ledgard, S.F., and Clark, H. 2003. Revised nitrous oxide emissions from New Zealand agricultural soils: 1990–2001. Ministry of Agriculture & Forestry.

Keoleian, G.A., and T.A. Volk. 2005. Renewable energy from willow biomass crops: life cycle energy, environmental and economic performance. *Critical Reviews in Plant Sciences* 24(5): 385-406.

Kilpelainen, A., A. Alam, H. Strandman, et al. 2011. Life cycle assessment tool for estimating net CO2 exchange of forest production. *Global Change Biology Bioenergy* 3(6): 461-471.

Kim, M., B.A. McCarl, and B.C. Murray. 2008. Permanence discounting for land-based carbon sequestration. *Ecol. Econ*. 64(4): 763-769.

Klein-Marcuschamer, D., P. Oleskowicz-Popiel, B.A. Simmons, et al. 2012. The Challenge of Enzyme Cost in the Production of Lignocellulosic Biofuels. 109(4): 1083-1087.

Knoll, K., West, B., Clark, W., Graves, R., Orban, J., Przesmitzki, S., and Theiss, T. 2009. Effects of Intermediate Ethanol Blends on Legacy Vehicles and Small Non-Road Engines. National Renewable Energy Laboratory. Rep. NREL/TP-540-43543.

Kull, S.J., Kurz, W.A., Rampley, G.J., Banfield, G.E., Scivatcheva, R.K., and Apps, M.J. 2007. Operational-Scale Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3). Rep. Version 1.0: User's Guide.

Kumabe, K., T. Hanaoka, S. Fujimoto, et al. 2007. Co-gasification of woody biomass and coal with air and steam. *Fuel* 86(5-6): 684-689.

Kurz, W., and M. Apps. 2006. Developing Canada's National Forest Carbon Monitoring, Accounting and Reporting System to Meet the Reporting Requirements of the Kyoto Protocol. *Mitigation Adapt. Strat. Global Change* 11(1): 33-43.

Kurz, W.A., S.J. Beukema, and M.J. Apps. 1996. Estimation of root biomass and dynamics for the carbon budget model of the Canadian forest sector. *Can. J. For. Res.* 26(11): 1973-1979.

Kurz, W.A., C.C. Dymond, T.M. White, et al. 2009. CBM-CFS3: A model of carbon-dynamics in forestry and land-use change implementing IPCC standards. *Ecol. Model.* 220(4): 480-504.

Laganiere, J., D.A. Angers, and D. Pare. 2010. Carbon accumulation in agricultural soils after afforestation: a meta-analysis. *Global Change Biology* 16(1): 439-453.

Lal, R. 2004. Carbon emission from farm operations. *Environ. Int.* 30(7): 981-990.

Larsen, J., M.Ø Haven, and L. Thirup. 2012. Inbicon makes lignocellulosic ethanol a commercial reality. *Biomass Bioenergy* 46(0): 36-45.

Larson, E.D. 2006. A review of life-cycle analysis studies on liquid biofuel systems for the transport sector. *Energy for Sustainable Development* 10(2): 109-126.

Laureysens, I., A. Pellis, J. Willems, et al. 2005. Growth and production of a short rotation coppice culture of poplar. III. Second rotation results. *Biomass and Bioenergy* 29(1): 10-21.

Lazarus, W. 2011. Machinery Cost Estimates. University of Minnesota Extension. http://faculty.apec.umn.edu/wlazarus/documents/machdata.pdf. Retrieved on: March 2012.

LeBlanc, A. 1999. Issues related to including forestry-based offsets in a GHG emissions trading system. *Environ. Sci. & Policy* 2(2): 199-206.

Lewandowski, I., and A. Faaij. 2006. Steps towards the development of a certification system for sustainable bio-energy trade. *Biomass and Bioenergy* 30(2): 83-104.

Li, Z., W.A. Kurz, M.J. Apps, et al. 2003. Belowground biomass dynamics in the Carbon Budget Model of the Canadian Forest Sector: Recent improvements and implications for the estimation of NPP and NEP. *Can. J. For. Res.* 33: 126-136.

Lindeman, R.H., Merenda, P.F., and Gold, R.Z. 1980. Introduction to Bivariate and Multivariate Analysis. Scott, Foresman, Glenview IL.

Lippke, B., J. Wilson, J. Perez-Garcia, et al. 2004. CORRIM: Life-Cycle Environmental Performance of Renewable Building Materials. *For. Prod. J.* 54(6): 8-19.

Liska, A.J., and K.G. Cassman. 2008. Towards standardization of life-cycle metrics for biofuels: greenhouse gas emissions mitigation and net energy yield. *Journal of Biobased Materials and Bioenergy* 2: 187-203.

Liska, A.J., and R.K. Perrin. 2009. Indirect land use emissions in the life cycle of biofuels: regulations *vs* science. *Biofuels, Bioproducts and Biorefining* 3(3): 318-328.

Liska, A.J., H.S. Yang, V.R. Bremer, et al. 2009. Improvements in Life Cycle Energy Efficiency and Greenhouse Gas Emissions of Corn-Ethanol. *J. Ind. Ecol.* 13(1): 58-74.

Lynd, L.R., P.J. Weimer, W.H. van Zyl, et al. 2002. Microbial Cellulose Utilization: Fundamentals and Biotechnology. *Microbiology and Molecular Biology Reviews* 66(3): 506-577.

Lynd, L.R., J.H. Cushman, R.J. Nichols, et al. 1991. Fuel Ethanol from Cellulosic Biomass. *Science* 251(4999): 1318-1323.

Lynd, L., R. Elamder, and C. Wyman. 1996. Likely features and costs of mature biomass ethanol technology. *Appl. Biochem. Biotechnol.* 57-58(1): 741-761.

Macdonald, D., J. Donner, and A. Nikiforuk. 1997. Full fuel cycle emission analysis for electric power generation options and its application in a market-based economy. *Energy Conversion and Management* 38(Supplement 1): S601-S606.

MacLean, H.L., and S. Spatari. 2009. The contribution of enzymes and process chemicals to the life cycle of ethanol. *Environmental Research Letters* 4

Andrew O. Makhorin. 2011. GNU Linear Programming Kit version 4.45. http://www.gnu.org/software/glpk/. Retrieved on: January 2013.

Maness, T.C. 2009. Forest Management and Climate Change Mitigation: Good Policy Requires Careful Thought. *J. For.* 107(April/May): 119-124.

Maniatis, K., D. Chiaramonti, and P. Thornley. 2012. Framework and perspectives of industrial lignocellulosic ethanol deployment: Introduction to the 1st International Conference on Lignocellulosic Ethanol. *Biomass Bioenergy* 46(0): 1-4.

Mann, M., and P. Spath. 2001. A life cycle assessment of biomass cofiring in a coal-fired power plant. *Clean Technologies and Environmental Policy* 3(2): 81-91.

Mao, R., D. Zeng, Y. Hu, L. Li, and D. Yang. 2010. Soil organic carbon and nitrogen stocks in an age-sequence of poplar stands planted on marginal agricultural land in Northeast China. Plant and Soil Springer Netherlands, **332:** 277-287.

Marland, G., and B. Schlamadinger. 1997. Forests for carbon sequestration or fossil fuel substitution? A sensitivity analysis. *Biomass Bioenergy* 13(6): 389-397.

Marshall, L. 2009. Biofuels and the Time Value of Carbon: Recommendations for GHG Accounting Protocols. World Resources Institute,. Washington, D.C. http://pdf.wri.org//working_papers/time_value_of_carbon.pdf. Retrieved on: October 2012.

Martin, C.W. 1988. Soil Disturbance by Logging in New England-Review and Management Recommendations. *Northern Journal of Applied Forestry* 5(1): 30-34.

Martin, N., Anglani, N., Einstein, D., Khrushch, M., Worrell, E., and Price, L.K. 2000. Opportunities to improve energy efficiency and reduce greenhouse gas emissions in the U.S. pulp and paper industry. Lawrence Berkeley National Laboratory. http://escholarship.org/uc/item/31b2f7bd. Retrieved on: August 2013.

Masera, O.R., J.F. Garza-Caligaris, M. Kanninen, et al. 2003. Modeling carbon sequestration in afforestation, agroforestry and forest management projects: the CO2FIX V.2 approach. *Ecol. Model*. 164(2-3): 177-199.

Mathews, J.A., and H. Tan. 2009. Biofuels and indirect land use change effects: the debate continues. *Biofuels, Bioproducts and Biorefining* 3(3): 305-317.

Matthews, H.S., C.T. Hendrickson, and C.L. Weber. 2008. The Importance of Carbon Footprint Estimation Boundaries. *Environ. Sci. Technol*. 42(16): 5839-5842.

Matthews, R.W. 2001. Modelling of energy and carbon budgets of wood fuel coppice systems. *Biomass and Bioenergy* 21(1): 1-19.

McAloon, A., Taylor, F., Yee, W., Ibsen, K., and Wooley, R. 2000. Determining the cost of producing ethanol from corn starch and lignocellulosic feedstocks. NREL - National Renewable Energy Laboratory. Rep. NREL/TP-580-28893.Golden, CO.

McAuliffe, P. 2011. Short Rotation Culture of Woody Crops for Bio-energy Production in B.C. P. McAuliffe and Associates Ltd. Victoria, BC.

McKechnie, J., S. Colombo, J. Chen, et al. 2011. Forest Bioenergy or Forest Carbon? Assessing Trade-Offs in Greenhouse Gas Mitigation with Wood-Based Fuels. *Environ. Sci. Technol.* 45(2): 789-795.

McKenney, D.W., D. Yemshanov, G. Fox, et al. 2004. Cost estimates for carbon sequestration from fast growing poplar plantations in Canada. *Forest Policy and Economics* 6(3-4): 345-358.

Melillo, J.M., J.M. Reilly, D.W. Kicklighter, et al. 2009. Indirect Emissions from Biofuels: How Important? *Science*

Miller, R.O., and Bender, B.A. 2008. Growth and yield of poplar and willow hybrids in the central upper peninsula of Michigan. *In Edited by* Anonymous

Miner, R. 2006. The 100-Year Method for Forecasting Carbon Sequestration in Forest Products in Use. *Mitigation and Adaptation Strategies for Global Change*

Mitchell, C.P., E.A. Stevens, and M.P. Watters. 1999. Short-rotation forestry – operations, productivity and costs based on experience gained in the UK. *For. Ecol. Manage.* 121(1-2): 123-136.

Montgomery, R. 2004. Development of biobased products. *Bioresour. Technol.* 91(1): 1-29.

Mosier, N., C. Wyman, B. Dale, et al. 2005. Features of promising technologies for pretreatment of lignocellulosic biomass. *Bioresour. Technol.* 96(6): 673-686.

Murray, L.D., L.B. Best, T.J. Jacobsen, et al. 2003. Potential effects on grassland birds of converting marginal cropland to switchgrass biomass production. *Biomass Bioenergy* 25(2): 167-175.

National Research Council. 2009. Liquid Transportation Fuels from Coal and Biomass: Technological Status, Costs, and Environmental Impacts. The National Academies Press. Washington, DC.

Nave, L.E., E.D. Vance, C.W. Swanston, et al. 2010. Harvest impacts on soil carbon storage in temperate forests. *For. Ecol. Manage.* 259(5): 857-866.

Neilson, E.T., D.A. MacLean, F.R. Meng, et al. 2008. Optimal on- and off-site forest carbon sequestration under existing timber supply constraints in northern New Brunswick. *Canadian Journal of Forest Research* 38(11): 2784-2796.

Neilson, E.T., D.A. MacLean, P.A. Arp, et al. 2006. Modeling carbon sequestration with CO sub(2)Fix and a timber supply model for use in forest management planning. *Canadian Journal of Soil Science* 86(2): 219-233.

Neilson, E.T., D.A. MacLean, F.-. Meng, et al. 2007. Spatial distribution of carbon in natural and managed stands in an industrial forest in New Brunswick, Canada. *For. Ecol. Manage.* 253(1-3): 148-160.

New Holland North America, Inc. 2002. New Holland Self-Propelled Forage Harvester Specifications. New Holland North America, Inc. http://www.twinfallstractor-imp.com/pics/fxspspecs.pdf. Retrieved on: January 2012.

O'Hare, M., R.J. Plevin, J.I. Martin, et al. 2009. Proper accounting for time increases crop-based biofuels' greenhouse gas deficit versus petroleum. *Environmental Research Letters* 4(2): 024001.

Olander, L. 2008. Designing Offsets Policy for the U.S. Nicholas Institute for Environmental Policy Solutions. Rep. NI R 08-01.Duke University, Durham N.C.

O'Laughlin, J. 2010. Accounting for Greenhouse Gas Emissions from Wood Bioenergy. Policy Analysis Group, University of Idaho, College of Natural Resources. Rep. Issue 31.

Paul, K.I., P.J. Polglase, and G.P. Richards. 2003. Predicted change in soil carbon following afforestation or reforestation, and analysis of controlling factors by linking a C accounting model (CAMFor) to models of forest growth (3PG), litter decomposition (GENDEC) and soil C turnover (RothC). *For. Ecol. Manage.* 177(1-3): 485-501.

Paul, K.I., P.J. Polglase, J.G. Nyakuengama, et al. 2002. Change in soil carbon following afforestation. *Forest Ecology and Management* 168(1-3): 241-257.

Pearson, T.R.H., Brown, S.L., and Birdsey, R.A. 2007. Measurement guidelines for the sequestration of forest carbon. U.S. Department of Agriculture, Forest Service, Northern Research Statio. Rep. Gen. Tech. Rep. NRS-18.Newtown Square, PA.

Peichl, M., N. Thevathasan, A. Gordon, et al. 2006. Carbon Sequestration Potentials in Temperate Tree-Based Intercropping Systems, Southern Ontario, Canada. *Agroforestry Systems* 66(3): 243-257.

Penman, J., Gytarsky, M., Hiraishi, T., Krug, T., Kruger, D., Pipatti, R., Buendia, R., Miwa, K., Ngara, T., Tanabe, K., and Wagner, F. 2003. Good Practice Guidance for Land Use, Land-Use Change and Forestry. Institute for Global Environmental Strategies. Hayama, Kanagawa Japan. Available at: Available at http://www.ipcc-nggip.iges.or.jp. Retrieved on: June 2013.

Perez-Garcia, J., B. Lippke, J. Comnick, et al. 2005. An Assessment of Carbon Pools, Storage, and Wood Products Market Substitution Using Life-Cycle Analysis Results. *Wood and Fiber Science* (37): 140-148.

Perlack, R.D., Wright, L.L., Turhollow, A.F., Graham, R.L., Stokes, B.J., and Erbach, D.C. 2005. Biomass as Feedstock for a Bioenergy and Bioproduction Industry: The Technical Feasibility of a Billion-Ton Annual Supply.

Phillips, S.D. 2007. Technoeconomic Analysis of a Lignocellulosic Biomass Indirect Gasification Process To Make Ethanol via Mixed Alcohols Synthesis. *Ind Eng Chem Res* 46(26): 8887-8897.

Piccolo, C., and F. Bezzo. 2009. A techno-economic comparison between two technologies for bioethanol production from lignocellulose. *Biomass and Bioenergy* 33(3): 478-491.

Pimentel, D. 2003. Ethanol Fuels: Energy Balance, Economics, and Environmental Impacts Are Negative. *Natural Resources Research* 12(2): 127-134.

PRé Consultants. SimaPro LCA Software. Available at: http://www.pre-sustainability.com/simapro-lca-software. Last updated: 2012. Accessed: 2013 January.

PricewaterhouseCoopers, L. 2009. Carbon Disclosure Project: Supply chain report 2009. Carbon Disclosure Project. London, UK.

Ptasinski, K.J., M.J. Prins, and A. Pierik. 2007. Exergetic evaluation of biomass gasification. *Energy* 32(4): 568-574.

Canada Queen's Printer. Farm Practices Protection (Right to Farm) Act. Available at: http://www.qp.gov.bc.ca/statreg/stat/F/96131_01.htm. Last updated: 1996. Accessed: 2009 December.

R Core Team. 2012. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing.

Rabl, A., A. Benoist, D. Dron, et al. 2007. How to account for CO2 emissions from biomass in an LCA. *The International Journal of Life Cycle Assessment* 12(5): 281.

Rafaschieri, A., M. Rapaccini, and G. Manfrida. 1999. Life Cycle Assessment of electricity production from poplar energy crops compared with conventional fossil fuels. *Energy Conversion and Management* 40(14): 1477-1493.

Ragauskas, A.J., C.K. Williams, B.H. Davison, et al. 2006. The Path Forward for Biofuels and Biomaterials. *Science* 311(5760): 484-489.

Ramlal, E., D. Yemshanov, G. Fox, et al. 2009. A bioeconomic model of afforestation in Southern Ontario: Integration of fiber, carbon and municipal biosolids values. *J. Environ. Manage.* 90(5): 1833-1843.

Revell, L.E., G.E. Bodeker, P.E. Huck, et al. 2012. Impacts of the production and consumption of biofuels on stratospheric ozone. *Geophys. Res. Lett.* 39(10)

Robertson, G.P., V.H. Dale, O.C. Doering, et al. 2008. Sustainable Biofuels Redux. *Science* 322(5898): 49-50.

Robinson, A.L., J.S. Rhodes, and D.W. Keith. 2003. Assessment of Potential Carbon Dioxide Reductions Due to Biomass−Coal Cofiring in the United States. *Environ. Sci. Technol.* 37(22): 5081-5089.

Rose, D., Ferguson, K., Lothner, D.C., and Zavitkovski, J. 1981. An economic and energy analysis of poplar intensive cultures in the Lake States. U.S. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station. Rep. Research Paper NC-196.St. Paul, MN.

Rothausen, S.G.S.A., and D. Conway. 2011. Greenhouse-gas emissions from energy use in the water sector. *Nature Clim. Change* 1(4): 210-219.

Roundtable on Sustainable Biofuels. 2011. RSB GHG calculation methodology. Rep. RSB-STD-01-003-01 (Version 2.0). Retrieved on: June 2012.

Roundtable on Sustainable Biofuels. 2011. Consolidated RSB EU RED Principles & Criteria. Rep. [RSB-STD-11-001-01-001 (Version 2.0)].Lausanne, Switzerland. http://cgse.epfl.ch/webdav/site/cgse/users/171495/public/RSB-brochure-eng.pdf;. Retrieved on: June 2012.

Roundtable on Sustainable Biofuels. 2010. RSB Principles and Criteria for Sustainable Biofuels Production. Version 2.0. Ecole Polytechnique Fédérale de Lausanne - Energy Center. Rep. [RSB-STD-01-001 (Version 2.0)].Lausanne, Switzerland. http://cgse.epfl.ch/webdav/site/cgse/users/171495/public/RSB-brochure-eng.pdf. Retrieved on: August 2011.

Rubin, E.M. 2008. Genomics of cellulosic biofuels. *Nature* 454(7206): 841-845.

Ruddell, S., R. Sampson, M. Smith, et al. 2007. The Role for Sustainably Managed Forests in Climate Change Mitigation. *Journal of Forestry* 105(6): 314-319.

Sambo, S.M. 2002. Fuel consumption for ground-based harvesting systems in western Canada. *Advantage* 3(29)

Sampson, N., Ruddell, S., and Smith, M. 2007. Managed Forests in Climate Change Policy: Program Design Elements. Society of American Foresters. http://www.forestcarbonstandards.org/ManagedForests_FinalDec14_2007.pdf. Retrieved on: September 2012.

Samson, R., Girouard, P., Zan, C., Mehdi, B., Martin, R., and Henning, J. 1999. The Implications of Growing Short-Rotation Tree Species for Carbon Sequestration in Canada. R.E.A.P. Canada. Rep. 23103-8-0253/N.Ste. Anne de Bellevue, QC.

Sartori, F., R. Lal, M.H. Ebinger, et al. 2007. Tree species and wood ash affect soil in Michigan's Upper Peninsula. *Plant & Soil* 298(1): 125-144.

Sassner, P., M. Galbe, and G. Zacchi. 2008. Techno-economic evaluation of bioethanol production from three different lignocellulosic materials. *Biomass Bioenergy* 32(5): 422-430.

Scharlemann, J.P.W., and W.F. Laurance. 2008. ENVIRONMENTAL SCIENCE: How Green Are Biofuels? *Science* 319(5859): 43-44.

Schelhaas, M.J., van Esch, P.W., Groen, T.A., de Jong, B.H.J., Kanninen, M., Liski, J., Masera, O., Mohren, G.M.J., Nabuurs, G.J., Palosuo, T., Pedroni, L., Vallejo, A., and Vilen, T. 2004. CO2FIX V 3.1 - A modelling framework for quantifying carbon sequestration in forest ecosystems. Rep. Alterra Report 1068.Wageningen.

Schlamadinger, B., and G. Marland. 1996. Full fuel cycle carbon balances of bioenergy and forestry options. *Energy Conversion and Management* 37(6-8): 813-818.

Schlamadinger, B., and G. Marland. 1996. The role of forest and bioenergy strategies in the global carbon cycle. *Biomass Bioenergy* 10(5-6): 275-300.

Schmer, M.R., K.P. Vogel, R.B. Mitchell, et al. 2008. Net energy of cellulosic ethanol from switchgrass. *Proc. Nat. Acad. Sci. USA* 105(2): 464-469.

Schmidt, M.G., S.E. Macdonald, and R.L. Rothwell. 1996. Impacts of harvesting and mechanical site preparation on soil chemical properties of mixed-wood boreal forest sites in Alberta. *Canadian Journal of Soil Science* 76: 531-540.

Schulze, E., C. Körner, B.E. Law, et al. 2012. Large-scale bioenergy from additional harvest of forest biomass is neither sustainable nor greenhouse gas neutral. *CGB Bioenergy* 4(6): 611-616.

Scown, C.D., W.W. Nazaroff, U. Mishra, et al. 2012. Lifecycle greenhouse gas implications of US national scenarios for cellulosic ethanol production. *Environmental Research Letters* 7(1): 014011.

Searchinger, T., R. Heimlich, R.A. Houghton, et al. 2008. Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land-Use Change. *Science* 319(5867): 1238-1240.

Searchinger, T.D., S.P. Hamburg, J. Melillo, et al. 2009. Fixing a Critical Climate Accounting Error. *Science* 326(5952): 527-528.

Seely, B., C. Welham, and H. Kimmins. 2002. Carbon sequestration in a boreal forest ecosystem: results from the ecosystem simulation model, FORECAST. *Forest Ecology and Management* 69(1-2): 123-135.

Semelsberger, T.A., R.L. Borup, and H.L. Greene. 2006. Dimethyl ether (DME) as an alternative fuel. *J. Power Sources* 156(2): 497-511.

Seo, J.S., H. Chong, H.S. Park, et al. 2004. The genome sequence of the ethanologenic bacterium Zymomonas mobilis ZM4. *Nature Biotechnology* 23: 63-68.

J. Sessions. 2010. Personal Communication. College of Forestry, Oregon State University.

SGA Energy Ltd. 2000. Emission Factors and Uncertainties for CH4 & N20 from Fuel Combustion.

Shapouri, H., Duffield, J.A., and Wang, M. 2002. The Energy Balance of Ethanol: An Update. U.S. Department of Agriculture. Rep. Report No. 814.Washington, D.C.

Shaw, C.H., Bhatti, J.S., and Sabourin, K.J. 2005. An ecosystem carbon database for Canadian forests. Canadian Forest Service, Northern Forestry Centre. Rep. NOR-X-403.Edmonton, AB.

Sheehan, J., A. Aden, K. Paustian, et al. 2004. Energy and Environmental Aspects of Using Corn Stover for Fuel Ethanol. *Journal of Industrial Ecology* 7(3/4): 117-146.

Singh, A., D. Pant, N.E. Korres, et al. 2010. Key issues in life cycle assessment of ethanol production from lignocellulosic biomass: Challenges and perspectives. *Bioresour. Technol.* 101(13): 5003-5012.

Smith, J.E., and Heath, L.S. 2006. Methods for Calculating Forest Ecosystem and Harvested Carbon with Standard Estimates for Forest Types of the United States. USDA Forest Service. Rep. General Technical Report NE-343.Newton Square, PA.

Smith, J.E., Heath, L.S., and Nichols, M.C. 2007. U.S. Forest Carbon Calculation Tool: Forest-Land Carbon Stocks and Net Annual Stock Change. U.S. Department of Agriculture, Forest Service, Northern Research Station. Rep. Gen. Tech. Rep. NRS-13.Newtown Square, PA. http://nrs.fs.fed.us/pubs/2394. Retrieved on: August 2012.

Smith, K.A., and T.D. Searchinger. 2012. Crop-based biofuels and associated environmental concerns.

So, K., and R. Brown. 1999. Economic analysis of selected lignocellulose-to-ethanol conversion technologies. *Appl. Biochem. Biotechnol.* 79(1): 633-640.

Soil Classification Working Group. 1998. The Canadian system of soil classification. Agriculture and Agri-Food Canada Publ. 1646 (revised) ed. NRC Research Press, Ottawa, Ontario.

Somerville, C., H. Youngs, C. Taylor, et al. 2010. Feedstocks for Lignocellulosic Biofuels. *Science* 329(5993): 790-792.

Spatari, S., D.M. Bagley, and H.L. MacLean. 2010. Life cycle evaluation of emerging lignocellulosic ethanol conversion technologies. *Bioresour. Technol.* 101(2): 654-667.

Spatari, S., Y. Zhang, and H.L. MacLean. 2005. Life Cycle Assessment of Switchgrass- and Corn Stover-Derived Ethanol-Fueled Automobiles. *Environ. Sci. Technol.* 39(24): 9750-9758.

Spath, P., and Dayton, D.C. 2003. Preliminary screening – Technical and economic assessment of synthesis gas to fuels and chemicals with emphasis on the potential for biomass-derived syngas. Rep. A925634.

Spath, P., Aden, A., Eggeman, T., Ringer, M., Wallace, B., and Jechura, J. 2005. Biomass to Hydrogen Production Detailed Design and Economics Utilizing the Battelle Columbus Laboratory Indirectly-Heated Gasifier. National Renewable Energy Laboratory. Rep. NREL/TP-510-37408.Golden, CO.

Standish, J.T., Manning, G.H., and Demaerschalk, J.P. 1985. Development of biomass equations for British Columbia tree species. Pacific Forest Research Centre. Rep. Info. Rep. BC-X-264.Victoria, B.C.

B. J. Stanton. 2013. Personal Communication.

Stanturf, J.A., van Oosten, C., Netzer, D.A., Coleman, M.D., and Portwood, C.J. 2001. Ecology and silviculture of poplar plantations. *In* Poplar Culture in North America. National Research Council, Ottawa, Canada pp. 153-206.

Stephenson, A.L., H. von Blottnitz, A.C. Brent, et al. 2010. Global Warming Potential and Fossil-Energy Requirements of Biodiesel Production Scenarios in South Africa. *Energy Fuels* 24(4): 2489-2499.

Stone, K., and L. Lynd. 1995. Analysis of internal and external energy flows associated with projected process improvements in biomass ethanol production. *Appl. Biochem. Biotechnol.* 51-52(1): 569-584.

Sullivan, J.L., Burnham, J.A., and Wang, M. 2010. Energy-Consumption and Carbon-Emission Analysis of Vehicle and Component Manufacturing. Argonne National Laboratory. Rep. ANL/ESD/10-6.

Tembo, G., F.M. Epplin, and R.L. Huhnke. 2003. Integrative investment appraisal of a lignocellulosic biomass-to-ethanol industry. *Journal of Agricultural and Resource Economics* 28(3): 611-633.

The University of British Columbia. Structure of Theses and Dissertations. Available at: https://www.grad.ubc.ca/current-students/dissertation-thesis-preparation/structure-theses-dissertations. Last updated: 2013. Accessed: 2014 January.

Thorsell, S., F.M. Epplin, R.L. Huhnke, et al. 2004. Economics of a coordinated biorefinery feedstock harvest system: lignocellulosic biomass harvest cost. *Biomass and Bioenergy* 27(4): 327-337.

TIAX LLC. 2007. Full fuel cycle assessment: well-to-wheels energy inputs, emissions, and water impacts. California Energy Commission. Rep. CEC-600-2007-004-REV.

Tilman, D., J. Hill, and C. Lehman. 2006. Carbon-Negative Biofuels from Low-Input High-Diversity Grassland Biomass. *Science* 314(5805): 1598-1600.

Tilman, D., R. Socolow, J.A. Foley, et al. 2009. Beneficial Biofuels--The Food, Energy, and Environment Trilemma. *Science* 325(5938): 270-271.

Timberline Natural Resource Group Ltd. 2009. Sibac Princeton Fibre use and Fibre Supply Study. The Southern Interior Beetle Action Coalition. Rep. BC0609098.

Tonn, B., and G. Marland. 2007. Carbon sequestration in wood products: a method for attribution to multiple parties. *Environmental Science & Policy* 10(2): 162-168.

Tubby, I., and Armstrong, A. 2002. Establishment and management of short rotation coppice - Practice Note. UK Forestry Commission. Edinburgh, UK.

Tyner, W.E., Taheripour, F., Zhuang, Q., Birur, D., and Baldos, U. 2010. Land use changes and Consequent CO2 Emissions due to US Corn Ethanol Production: A Comprehensive Analysis. Department of Agricultural Economics, Purdue University. Rep. GTAP Resource 3288, Final Report, revised.

U.S. Department of Energy. 2010. CO2 stationary source emission estimation methodologies summary. Appendix A. http://www.netl.doe.gov/technologies/carbon_seq/refshelf/atlasIII/2010AtlasIII_AppendixA.pdf. Retrieved on: May 2012.

U.S. Department of Energy. Theoretical Ethanol Yield Calculator. Available at: http://www1.eere.energy.gov/biomass/ethanol_yield_calculator.html. Last updated: 2009. Accessed: 2012 February.

U.S. DOE-NETL. 2008. Development of Baseline Data and Analysis of Life Cycle Greenhouse Gas Emissions of Petroleum-Based Fuels. US National Energy Technology Laboratory. Rep. DOE/NETL-2009/1346.Available at: Available at http://www.netl.doe.gov/energy-analyses/refshelf/results.asp?ptype=Models/Tools [Verified on May 30, 2010].

U.S. EPA. 2011. Greenhouse Gas Emissions from a Typical Passenger Vehicle. Office of Transportation and Air Quality. Rep. EPA-420-F-11-041.http://www.epa.gov/otaq/climate/documents/420f11041.pdf. Retrieved on: July 2012.

U.S. EPA. 2010. Regulation of Fuels and Fuel Additives: Modifications to Renewable Fuel Standard Program (RFS2); Final Rule.

U.S. National Renewable Energy Laboratory. Biomass Feedstock Composition and Property Database. Available at: http://www.afdc.energy.gov/biomass/progs/search1.cgi. Last updated: 2004. Accessed: 2012 November.

UK DEFRA (Department for Environment, Food and Rural Affairs). 2004. Growing Short Rotation Coppice - Best Practice Guidelines. Rep. PB 7135.

US EPA. 2008. Climate leaders greenhouse gas inventory protocol offset project methodology for project type: reforestation/afforestation. US Environmental Protection Agency, Office of Athmospheric Programs, Climate Protection Partnerships Division/Climate Change Division. Rep. EPA400-S-08-007.http://epa.gov/climatechange/emissions/. Retrieved on: October 2012.

van Dam, J., M. Junginger, A. Faaji, et al. 2008. Overview of recent developments in sustainable biomass certification. *Biomass and Bioenergy* 32(8): 749-780.

van Kooten, G.C., E. Krcmar-Nozic, B. Stennes, et al. 1999. Economics of fossil fuel substitution and wood product sinks when trees are planted to sequester carbon on agricultural lands in western Canada. *Can. J. For. Res.* 29(11): 1669-1678.

van Kooten, G.C. 2000. Economic Dynamics of Tree Planting for Carbon Uptake on Marginal Agricultural Lands. *Can. J. Agric. Econ.* 48(1): 51-65.

van Oosten, C. 2008. Activities related to poplar and willow cultivation and utilization in Canada. Poplar Council of Canada. Edmonton, AB.

van Oosten, C. 2006. Hybrid Poplar Crop Manual for the Prairie Provinces - Version 1. Saskatchewan Forest Centre. Prince Albert, SK.

van Oosten, C. 2006. Crop Density for Hybrid Poplar in the Prairie Provinces. Saskatchewan Forest Centre. Rep. 200501.Prince Albert, SK.

Verdonk, M., C. Dieperink, and A.P.C. Faaij. 2007. Governance of the emerging bio-energy markets. *Energy Policy* 35(7): 3909-3924.

Vesterdal, L., E. Ritter, and P. Gundersen. 2002. Change in soil organic carbon following afforestation of former arable land. *For. Ecol. Manage.* 169(1–2): 137-147.

Viikari, L., J. Vehmaanperä, and A. Koivula. 2012. Lignocellulosic ethanol: From science to industry. *Biomass Bioenergy* 46(0): 13-24.

von Blottnitz, H., and M.A. Curran. 2007. A review of assessments conducted on bio-ethanol as a transportation fuel from a net energy, greenhouse gas, and environmental life cycle perspective. *Journal of Cleaner Production* 15(7): 607-619.

Wang, M., Wu, Y., and Elgowainy, A. 2007. Operating Manual for GREET: Version 1.7. Argonne National Laboratory. Rep. ANL/ESD/05-3.

Wang, T., J. Chang, and P. Lv. 2005. Synthesis Gas Production via Biomass Catalytic Gasification with Addition of Biogas. *Energy Fuels* 19(2): 637-644.

WBCSD. 2004. Mobility 2030: Meeting the challenges to sustainability. World Business Council for Sustainable Development. Conches-Geneva.

WBGU. 2008.
World in Transition: Future Bioenergy and Sustainable Land Use. Summary for Policy-Makers. German Advisory Council on Global Change. Berlin, Germany.

Werner, F., and B. Nebel. 2007. Wood & other renewable resources. *The International Journal of Life Cycle Assessment* 12(7): 462-463.

Whitaker, J., K.E. Ludley, R.L. Rowe, et al. 2010. Sources of variability in greenhouse gas and energy balances for biofuel production: a systematic review. *CGB Bioenergy* 2: 99-112.

T. J. R. White, S. W. J. Dominy and D. J. Allen. Review of Best Practices for Tree Planting on Marginal Lands in Ontario. Available at: http://cfs.nrcan.gc.ca/subsite/glfc-tree-planting. Last updated: 2010. Accessed: 2011 December.

White, E.H., Abrahamson, L.P., Kopp, R.F., Nowak, C.A., and Zsuffa, L. 1991. Bioenergy plantations in northeastern North America. *In* Energy from Biomass and Wastes, *Edited by* Anonymous pp. 273-283.

Wibe, S. 2012. Carbon dioxide emissions from wood fuels in Sweden 1980–2100. *Journal of Forest Economics* 12(2): 123-130.

Wicke, B., V. Dornburg, M. Junginger, et al. 2008. Different palm oil production systems for energy purposes and their greenhouse gas implications. *Biomass Bioenergy* 32(12): 1322-1337.

Wiedmann, T., and Minx, J. 2007. A Definition of "Carbon Footprint". ISA UK Research & Consulting. Rep. ISA UK Research Report 07-01.Durham, U.K.

Wiloso, E.I., R. Heijungs, and G.R. de Snoo. 2012. LCA of second generation bioethanol: A review and some issues to be resolved for good LCA practice. *Renewable and Sustainable Energy Reviews* 16(7): 5295-5308.

Wise, M., K. Calvin, A. Thomson, et al. 2009. Implications of Limiting CO2 Concentrations for Land Use and Energy. *Science* 324(5931): 1183-1186.

Wooley, R., and Putsche, V. 1996. Development of an ASPEN PLUS physical property database for biofuels components. National Renewable Energy Laboratory. Rep. NREL/MP-425-20685.Golden, CO.

Wooley, R., Ruth, M., Sheehan, J., Ibsen, K., Majdeski, H., and Galvez, A. 1999. Lignocellulosic biomass to ethanol process design and economics utilizing co-current dilute acid pre-hydrolysis and enzymatic hydrolysis current and futuristic scenarios. National Renewable Energy Laboratory. Golden, CO. http://www.nrel.gov/docs/fy99osti/26157.pdf. Retrieved on: April 2012.

World Resources Institute. 2008. GHG Protocol Corporate Accounting and Reporting Standard (Corporate Standard) (Draft). World Resources Institute and World Business Council for Sustainable Development. http://www.ghgprotocol.org/files/ghgp/psp-draft-1.pdf. Retrieved on: January 2009.

Wu, M., Wang, M., and Huo, H. 2006. Fuel-Cycle Assessment of Selected Bioethanol Production Pathways in the United States. Argonne National Laboratory. Rep. ANL/ESD/06-7.

Wyman, C.E. 2003. Potential Synergies and Challenges in Refining Cellulosic Biomass to Fuels, Chemicals, and Power. *Biotechnol. Prog.* 19(2): 254-262.

Yacobucci, B.D., and Bracmort, K. 2010. Calculation of Lifecycle Greenhouse Gas Emissions for the Renewable Fuel Standard (RFS). US Congressional Research Service.

Yanai, R.D., W.S. Currie, and C.L. Goodale. 2003. Soil Carbon Dynamics after Forest Harvest: An Ecosystem Paradigm Reconsidered. *Ecosystems* 6(3): 197-212.

Young, S.B. 2003. Life cycle assessment in Canada. *The International Journal of Life Cycle Assessment* 8(6): 321-322.

Zah, R., Böni, H., Gauch, M., Hischier, R., Lehmann, M., and Wagner, P. 2007. Life cycle assessment of energy products: environmental impact assessment of biofuels. EMPA. St Gallen, Switzerland.

Zanchi, G., N. Pena, and N. Bird. 2012. Is woody bioenergy carbon neutral? A comparative assessment of emissions from consumption of woody bioenergy and fossil fuel. *GCB Bioenergy* 4(6): 761-772.

Zhang, J., T. Shangguan, and Z. Meng. 2010. Changes in soil carbon flux and carbon stock over a rotation of poplar plantations in northwest China. *Ecol. Res.* 26(1): 153-161.

Zhang, Y., S. Joshi, and H.L. MacLean. 2010. Can ethanol alone meet California's low carbon fuel standard? An evaluation of feedstock and conversion alternatives. *Environmental Research Letters* 5(1): 014002 (14pp).

Zwart, R.W.R., and H. Boerrigter. 2005. High Efficiency Co-production of Synthetic Natural Gas (SNG) and Fischer–Tropsch (FT) Transportation Fuels from Biomass. *Energy Fuels* 19(2): 591-597.

APPENDIX A. Biomass data

Table A.1. Biomass components distribution with age, as a proportion of total above-ground live biomass

	Stem	Bark	Branch	Foliage
Growth year	[g stem biomass/ g above-ground biomass]*	[g bark biomass / g above-ground biomass]	[g branch biomass / g above-ground biomass]	[g foliage biomass / g above-ground biomass]
1	0.448	0.209	0.230	0.112
2	0.519	0.170	0.225	0.086
3	0.563	0.145	0.222	0.070
4	0.588	0.131	0.221	0.060
5	0.603	0.123	0.220	0.055
6	0.612	0.118	0.219	0.051
7	0.617	0.115	0.219	0.049
8	0.621	0.113	0.218	0.048

* The sum of the values in each row is 1.

As expected, the biomass equations projected that the stem proportion increased as the trees mature, while the branch, bark and foliage proportions decreased with age.

Table A.2. Coarse and fine roots distribution with age, as a proportion of total above-ground live biomass

	Coarse roots	Fine roots
Growth year	[g coarse root biomass/ g above-ground biomass]	[g fine root biomass/ g above-ground biomass]
1	0.436	0.193
2	0.359	0.128
3	0.321	0.098
4	0.296	0.079
5	0.278	0.066
6	0.263	0.057
7	0.251	0.050
8	0.242	0.044

The costs in Table A.3 through Table A.5, expressed as $/ha/yr, were first calculated on a per hectare basis for each treatment (in some cases depending on the amount of biomass processed) and then divided by the number of years of the respective rotation specific for each treatment.

Table A.3. Costs by treatment for biomass production activities on a per hectare per year basis [$/ha/yr]

	Treatment type			
	S	Si	C	Ci
Cuttings/Rods	25.26	38.43	107.64	35.88
Planting labour	11.79	17.93	17.94	16.74
Herbicide prev. land use	18.53	--	9.88	--
Drip hose rollout	--	12.36	--	4.94
Ripping	4.63	6.18	2.47	6.59
Pre-emergent herbicide	10.81	14.41	5.77	5.77
Pre-plant herbicide	10.81	14.41	17.30	17.30
Backpack spray	6.18	20.59	3.29	8.24
Cultivation	64.87	--	69.19	--
Herbicide weed control	29.65	39.54	63.26	47.44
Pest control	--	61.78	--	61.78
Fertilization	18.53	34.59	16.47	50.24
Singling/Pruning	--	--	6.59	--
Drip hose mainten/rollup	--	17.30	--	23.06
Irrigation mgt., power, mainten.	--	426.26	--	370.66
Irrigation systems (capital)	--	473.62	--	189.45
Mgt cost	133.44	133.44	133.44	133.44
Mapping GIS	7.41	7.41	7.41	7.41
Other oper. expense (road maint., fire insurance)	37.07	37.07	37.07	37.07
Restoration	61.78	82.37	32.95	32.95

Table A.4. Costs for single stem treatments for harvesting and processing activities on a per hectare, per year basis [$/ha/yr]

	Treatment S	Treatment Si	Treatment C		
Rotation			1st	2nd	3rd
Felling, skidding, delimbing, loading	353.05	454.24	353.05	367.17	381.42
Grinding Mobile	26.26	34.01	26.60	27.66	28.77
Grinding Stationary	65.04	83.52	64.79	67.38	69.97

Table A.5. Costs for coppice irrigated treatment for harvesting and processing activities on a per hectare, per year basis [$/ha/yr]

	Treatment Ci				
Rotation	1st	2nd	3rd	4th	5th
Harvester, trailer, blower	223.64	228.12	230.35	232.59	234.82

APPENDIX B. Example of a simple model to illustrate the features of the C-BOS solver

To visually illustrate the features of the C-BOS solver we show model outputs for a simple model (Figure B.1). The project horizon was 13 years (with ethanol production starting in the 8th year), the minimum volume of ethanol needed was 10,000,000 l per year, one land unit was available (agricultural 1 non-irrigated) with a maximum area of 10,000 ha, and two biomass production treatments S and C were available. Treatment N (no-activity) with duration of one year was available in the first 4 years (*i.e.* years 0-3 in figure), and treatment I (idle) with duration of five years was available in the last 5 years (*i.e.* years 8-12 in figure).

Figure B.1 shows the list of all possible combinations of treatment sequences for the illustrative model. In this example there are 29 such possible sequences, shown as A_1 through A_29 in the figure. The no-activity treatments N are shown on a grey background with a grey font. The harvest years are highlighted with bold red font on grey background. For example, the A_3 sequence starts in year 0 with a single stem treatment S with a rotation age of 8 years (shown in the figure as $S[0:8]_0$). In the 8th year (*i.e.* year 7 in the figure) the treatment S is harvested (shown as $S[0:8]_7$). The following treatment is coppice non-irrigated C with a total rotation of 15 years, its first year is in year 8 of the project (shown as $C[8:15]_0$), and the first intermediate harvest is in year 5 of the treatment (year 12 of the project), shown as $C[8:15]_4$.

The 29 treatment sequences A_1 through A_29 also represent the possible parcels of land that the optimization model can choose from. When the mathematical optimization problem is solved, the model decides how many hectares to use for each treatment sequence, if any.

project year	0	1	2	3	4	5	6	7	8	9	10	11	12
Land unit lu_1 (agricultural1) [area <= 15000.00 ha]													
A_1	$S[0:8]_0$	$S[0:8]_1$	$S[0:8]_2$	$S[0:8]_3$	$S[0:8]_4$	$S[0:8]_5$	$S[0:8]_6$	**$S[0:8]_7$**	$I[8:6]_0$	$I[8:6]_1$	$I[8:6]_2$	$I[8:6]_3$	$I[8:6]_4$
A_2	$S[0:8]_0$	$S[0:8]_1$	$S[0:8]_2$	$S[0:8]_3$	$S[0:8]_4$	$S[0:8]_5$	$S[0:8]_6$	**$S[0:8]_7$**	$S[8:8]_0$	$S[8:8]_1$	$S[8:8]_2$	$S[8:8]_3$	$S[8:8]_4$
A_3	$S[0:8]_0$	$S[0:8]_1$	$S[0:8]_2$	$S[0:8]_3$	$S[0:8]_4$	$S[0:8]_5$	$S[0:8]_6$	**$S[0:8]_7$**	$C[8:15]_0$	$C[8:15]_1$	$C[8:15]_2$	$C[8:15]_3$	**$C[8:15]_4$**
A_4	$C[0:15]_0$	$C[0:15]_1$	$C[0:15]_2$	$C[0:15]_3$	**$C[0:15]_4$**	$C[0:15]_5$	$C[0:15]_6$	$C[0:15]_7$	$C[0:15]_8$	**$C[0:15]_9$**	$C[0:15]_{10}$	$C[0:15]_{11}$	$C[0:15]_{12}$
A_5	$N[0:1]_0$	$S[1:8]_0$	$S[1:8]_1$	$S[1:8]_2$	$S[1:8]_3$	$S[1:8]_4$	$S[1:8]_5$	$S[1:8]_6$	**$S[1:8]_7$**	$I[9:6]_0$	$I[9:6]_1$	$I[9:6]_2$	$I[9:6]_3$
A_6	$N[0:1]_0$	$S[1:8]_0$	$S[1:8]_1$	$S[1:8]_2$	$S[1:8]_3$	$S[1:8]_4$	$S[1:8]_5$	$S[1:8]_6$	**$S[1:8]_7$**	$S[9:8]_0$	$S[9:8]_1$	$S[9:8]_2$	$S[9:8]_3$
A_7	$N[0:1]_0$	$S[1:8]_0$	$S[1:8]_1$	$S[1:8]_2$	$S[1:8]_3$	$S[1:8]_4$	$S[1:8]_5$	$S[1:8]_6$	**$S[1:8]_7$**	$C[9:15]_0$	$C[9:15]_1$	$C[9:15]_2$	$C[9:15]_3$
A_8	$N[0:1]_0$	$C[1:15]_0$	$C[1:15]_1$	$C[1:15]_2$	$C[1:15]_3$	**$C[1:15]_4$**	$C[1:15]_5$	$C[1:15]_6$	$C[1:15]_7$	$C[1:15]_8$	**$C[1:15]_9$**	$C[1:15]_{10}$	$C[1:15]_{11}$
A_9	$N[0:1]_0$	$N[1:1]_0$	$S[2:8]_0$	$S[2:8]_1$	$S[2:8]_2$	$S[2:8]_3$	$S[2:8]_4$	$S[2:8]_5$	$S[2:8]_6$	**$S[2:8]_7$**	$I[10:6]_0$	$I[10:6]_1$	$I[10:6]_2$
A_10	$N[0:1]_0$	$N[1:1]_0$	$S[2:8]_0$	$S[2:8]_1$	$S[2:8]_2$	$S[2:8]_3$	$S[2:8]_4$	$S[2:8]_5$	$S[2:8]_6$	**$S[2:8]_7$**	$S[10:8]_0$	$S[10:8]_1$	$S[10:8]_2$
A_11	$N[0:1]_0$	$N[1:1]_0$	$S[2:8]_0$	$S[2:8]_1$	$S[2:8]_2$	$S[2:8]_3$	$S[2:8]_4$	$S[2:8]_5$	$S[2:8]_6$	**$S[2:8]_7$**	$C[10:15]_0$	$C[10:15]_1$	$C[10:15]_2$
A_12	$N[0:1]_0$	$N[1:1]_0$	$C[2:15]_0$	$C[2:15]_1$	$C[2:15]_2$	$C[2:15]_3$	**$C[2:15]_4$**	$C[2:15]_5$	$C[2:15]_6$	$C[2:15]_7$	$C[2:15]_8$	**$C[2:15]_9$**	$C[2:15]_{10}$
A_13	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$S[3:8]_0$	$S[3:8]_1$	$S[3:8]_2$	$S[3:8]_3$	$S[3:8]_4$	$S[3:8]_5$	$S[3:8]_6$	**$S[3:8]_7$**	$I[11:6]_0$	$I[11:6]_1$
A_14	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$S[3:8]_0$	$S[3:8]_1$	$S[3:8]_2$	$S[3:8]_3$	$S[3:8]_4$	$S[3:8]_5$	$S[3:8]_6$	**$S[3:8]_7$**	$S[11:8]_0$	$S[11:8]_1$
A_15	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$S[3:8]_0$	$S[3:8]_1$	$S[3:8]_2$	$S[3:8]_3$	$S[3:8]_4$	$S[3:8]_5$	$S[3:8]_6$	**$S[3:8]_7$**	$C[11:15]_0$	$C[11:15]_1$
A_16	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$C[3:15]_0$	$C[3:15]_1$	$C[3:15]_2$	$C[3:15]_3$	**$C[3:15]_4$**	$C[3:15]_5$	$C[3:15]_6$	$C[3:15]_7$	$C[3:15]_8$	**$C[3:15]_9$**
A_17	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$S[4:8]_0$	$S[4:8]_1$	$S[4:8]_2$	$S[4:8]_3$	$S[4:8]_4$	$S[4:8]_5$	$S[4:8]_6$	**$S[4:8]_7$**	$I[12:6]_0$
A_18	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$S[4:8]_0$	$S[4:8]_1$	$S[4:8]_2$	$S[4:8]_3$	$S[4:8]_4$	$S[4:8]_5$	$S[4:8]_6$	**$S[4:8]_7$**	$S[12:8]_0$
A_19	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$S[4:8]_0$	$S[4:8]_1$	$S[4:8]_2$	$S[4:8]_3$	$S[4:8]_4$	$S[4:8]_5$	$S[4:8]_6$	**$S[4:8]_7$**	$C[12:15]_0$
A_20	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$C[4:15]_0$	$C[4:15]_1$	$C[4:15]_2$	$C[4:15]_3$	**$C[4:15]_4$**	$C[4:15]_5$	$C[4:15]_6$	$C[4:15]_7$	$C[4:15]_8$
A_21	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$N[4:1]_0$	$S[5:8]_0$	$S[5:8]_1$	$S[5:8]_2$	$S[5:8]_3$	$S[5:8]_4$	$S[5:8]_5$	$S[5:8]_6$	**$S[5:8]_7$**
A_22	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$N[4:1]_0$	$C[5:15]_0$	$C[5:15]_1$	$C[5:15]_2$	$C[5:15]_3$	**$C[5:15]_4$**	$C[5:15]_5$	$C[5:15]_6$	$C[5:15]_7$
A_23	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$N[4:1]_0$	$N[5:1]_0$	$S[6:8]_0$	$S[6:8]_1$	$S[6:8]_2$	$S[6:8]_3$	$S[6:8]_4$	$S[6:8]_5$	$S[6:8]_6$
A_24	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$N[4:1]_0$	$N[5:1]_0$	$C[6:15]_0$	$C[6:15]_1$	$C[6:15]_2$	$C[6:15]_3$	**$C[6:15]_4$**	$C[6:15]_5$	$C[6:15]_6$
A_25	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$N[4:1]_0$	$N[5:1]_0$	$N[6:1]_0$	$S[7:8]_0$	$S[7:8]_1$	$S[7:8]_2$	$S[7:8]_3$	$S[7:8]_4$	$S[7:8]_5$
A_26	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$N[4:1]_0$	$N[5:1]_0$	$N[6:1]_0$	$C[7:15]_0$	$C[7:15]_1$	$C[7:15]_2$	$C[7:15]_3$	**$C[7:15]_4$**	$C[7:15]_5$
A_27	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$N[4:1]_0$	$N[5:1]_0$	$N[6:1]_0$	$N[7:1]_0$	$I[8:6]_0$	$I[8:6]_1$	$I[8:6]_2$	$I[8:6]_3$	$I[8:6]_4$
A_28	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$N[4:1]_0$	$N[5:1]_0$	$N[6:1]_0$	$N[7:1]_0$	$S[8:8]_0$	$S[8:8]_1$	$S[8:8]_2$	$S[8:8]_3$	$S[8:8]_4$
A_29	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$N[4:1]_0$	$N[5:1]_0$	$N[6:1]_0$	$N[7:1]_0$	$C[8:15]_0$	$C[8:15]_1$	$C[8:15]_2$	$C[8:15]_3$	**$C[8:15]_4$**
Constraints													
etoh range [10^6l]								10.00 2700.00	10.00 2700.00	10.00 2700.00	10.00 2700.00	10.00 2700.00	10.00 2700.00

Figure B.1. List of the 29 possible combinations of treatment sequences for the illustrative model

A graphical representation of the 29 possible treatment sequences is shown in Figure B.2. At year 0 there are three possible treatments: S, single stem non-irrigated, with a rotation age of 8 years (S[0:8]); C, coppice non-irrigated, with a total rotation age of 15 years (C[0:15]); and N, no-activity, with a rotation age of 1 year (N[0:1]). The same combination of three treatments is possible in all years until year 3. In year 4 only the S and C treatments are available, because treatment N was set to be available only in the first four years, *i.e.* years 0 to 3.

For simplicity, Figure B.2 shows only the treatment sequences that contain at least one treatment that ends (*i.e.* is harvested) within the planning horizon. For example, there are no treatments shown following C[0:15] in year 0 as this treatment ends (is harvested) in year 15, after the project end in year 13.

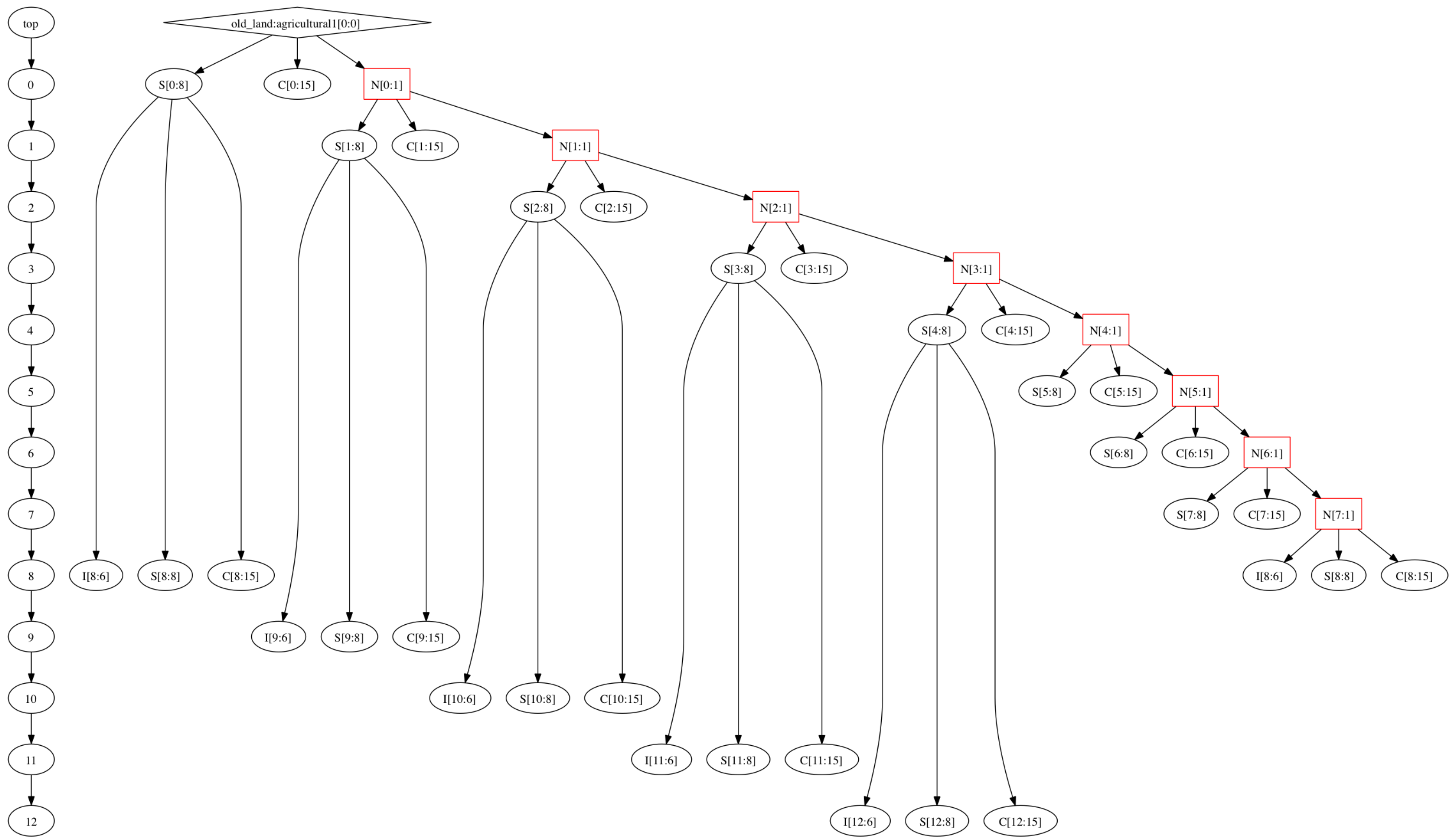

Figure B.2. Graphical representation of the 29 possible treatment sequences

The results from the simple illustrative model are shown in Figure B.3 and Table . Six land parcels (A_13, A_17, A_5, A_1, A_9, A_21; and their associated treatment sequences) are used by the model from the 29 total sequences available.

The minimum required ethanol volume of 10,000,000 l per year was produced as planned for 6 years, starting with the 8th year until the 13th year (this is shown as year 7 through 12 in the figure because the project start year is 0). A total area of 2,505.13 ha was used to produce the biomass needed per year.

We present as an example the calculations for the 8th year of the project horizon (shown as year 7 in Figure B.3). Parcel A_1 is harvested and produces 29,075,665.94 kg of stem biomass with treatment S (from: 13.35 t/ha * 8 years * 0.652 * 417.522 ha; where 13.35 t/ha is the biomass yield for above-ground biomass, and 0.652 is the proportion of stem biomass from the above-ground biomass). Parcel A_1 also produced 10,222,558.26 kg of stem biomass. The total biomass quantity converted to ethanol was 37,333,312.99 kg, representing 95% of the biomass available in the stem and branch biomass (*i.e.* 5% were not recovered due to mechanical losses), with conversion efficiency of 0.268 l EtOH/kg wood, resulting in 10,000,000 l EtOH.

solution ok?	true
total area used [ha]	2,505
site prep cost [$]	501,026
total rent cost [$]	6,437,925
total prod. cost [$]	17,738,314
total prod. cost/yr [$]	1,364,486
total prod. cost/yr/ha [$]	545
total transp. cost [$]	7,381,985
total transp. cost/yr [$]	567,845
total transp. cost/yr/ha [$]	227
objective value [$]	32,059,250

project year	0	1	2	3	4	5	6	7	8	9	10	11	12
Land unit lu_1 [soil_type: agricultural1; distance: 40 km; max_area: 15,000 ha; area used: 2,505 ha; dual_value: 0.00]													
A_13 [418 ha]	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$S[3:8]_0$	$S[3:8]_1$	$S[3:8]_2$	$S[3:8]_3$	$S[3:8]_4$	$S[3:8]_5$	$S[3:8]_6$	$S[3:8]_7$	$I[11:6]_0$	$I[11:6]_1$
A_17 [418 ha]	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$S[4:8]_0$	$S[4:8]_1$	$S[4:8]_2$	$S[4:8]_3$	$S[4:8]_4$	$S[4:8]_5$	$S[4:8]_6$	$S[4:8]_7$	$I[12:6]_0$
A_5 [418 ha]	$N[0:1]_0$	$S[1:8]_0$	$S[1:8]_1$	$S[1:8]_2$	$S[1:8]_3$	$S[1:8]_4$	$S[1:8]_5$	$S[1:8]_6$	$S[1:8]_7$	$I[9:6]_0$	$I[9:6]_1$	$I[9:6]_2$	$I[9:6]_3$
A_1 [418 ha]	$S[0:8]_0$	$S[0:8]_1$	$S[0:8]_2$	$S[0:8]_3$	$S[0:8]_4$	$S[0:8]_5$	$S[0:8]_6$	$S[0:8]_7$	$I[8:6]_0$	$I[8:6]_1$	$I[8:6]_2$	$I[8:6]_3$	$I[8:6]_4$
A_9 [418 ha]	$N[0:1]_0$	$N[1:1]_0$	$S[2:8]_0$	$S[2:8]_1$	$S[2:8]_2$	$S[2:8]_3$	$S[2:8]_4$	$S[2:8]_5$	$S[2:8]_6$	$S[2:8]_7$	$I[10:6]_0$	$I[10:6]_1$	$I[10:6]_2$
A_21 [418 ha]	$N[0:1]_0$	$N[1:1]_0$	$N[2:1]_0$	$N[3:1]_0$	$N[4:1]_0$	$S[5:8]_0$	$S[5:8]_1$	$S[5:8]_2$	$S[5:8]_3$	$S[5:8]_4$	$S[5:8]_5$	$S[5:8]_6$	$S[5:8]_7$
project year	0	1	2	3	4	5	6	7	8	9	10	11	12
EtOH output/year	0 0.00	0 0.00	0 0.00	0 0.00	0 0.00	0 0.00	0 0.00	10,000,000 0.53	10,000,000 0.53	10,000,000 0.53	10,000,000 0.53	10,000,000 0.53	10,000,000 0.53
Harvested stem/year	0	0	0	0	0	0	0	29,075,666	29,075,666	29,075,666	29,075,666	29,075,666	29,075,666
Harvested bark/year	0	0	0	0	0	0	0	5,276,399	5,276,399	5,276,399	5,276,399	5,276,399	5,276,399
Harvested branch/year	0	0	0	0	0	0	0	10,222,558	10,222,558	10,222,558	10,222,558	10,222,558	10,222,558
rent cost/year	495,225	495,225	495,225	495,225	495,225	495,225	495,225	495,225	495,225	495,225	495,225	495,225	495,225
prod cost/year	530,226	709,745	853,154	927,437	1,042,990	1,117,274	661,332	2,246,641	2,103,232	2,028,948	1,913,396	1,839,112	1,764,828
transportation cost/year	0	0	0	0	0	0	0	1,230,331	1,230,331	1,230,331	1,230,331	1,230,331	1,230,331

Figure B.3. Visual display of illustrative model results

The total project cost (the objective value of the optimization formulation) for the biomass production project over the planning horizon was $32,059,250.29, representing the sum of site preparation cost, land rent cost, production cost, and transport cost.

Table B.1. Illustrative model results

Total harvested biomass cost (objective value)	[$]	32,059,250.29
Site prep. cost	[$]	501,026.12
Land rent cost	[$]	6,437,924.97
Production cost	[$]	17,738,314.29
Transport cost	[$]	7,381,984.91
Total area used	[ha]	2,505.13
Total harvested biomass	[kg]	251,307,787.80
Total ethanol volume produced	[l]	60,000,000

APPENDIX C. Live biomass pools carbon dynamics

Figure C.1 through Figure C.5 show the carbon accumulation in live biomass pools over time, summed up by scenario for all the land parcels.

As expected, in scenarios 1A-4A where idle treatments were allowed, the carbon stock in live biomass pools is depleted in the last few years of the planning horizon due to abandoning the plantations. The gradual decrease of stored carbon in scenarios 3 and 4 between years 8-60 is due to the improvements in conversion efficiency (*i.e.* less biomass is needed to produce the same quantity of ethanol).

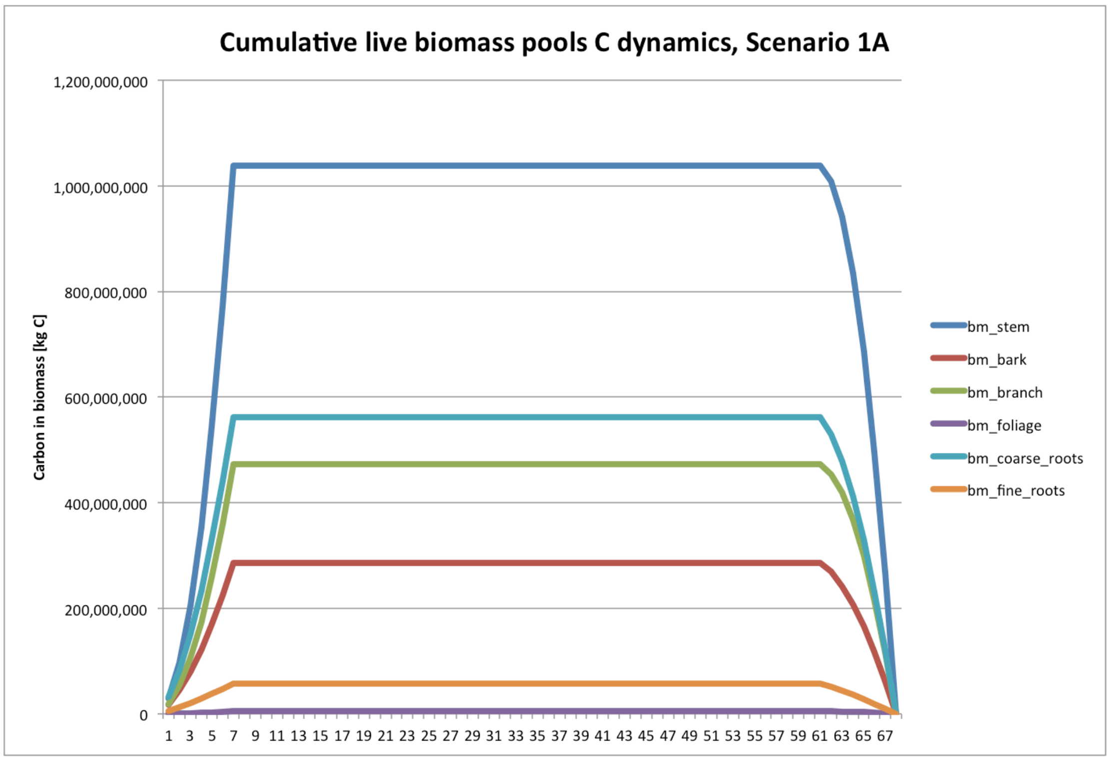

Figure C.1. Live biomass pools dynamics in all land parcels, scenario 1A

To illustrate how the live biomass pools carbon in each of the land parcels contributed to the total carbon for a scenario, Figure C.2 shows the carbon dynamics in the stem pool in each of the 8 land parcels for scenario 1A. The figure is a stacked graph and therefore the exterior perimeter of the graph has the exact same shape as the stem graph in Figure C.1. For scenario 1A it is relatively easy to show all the 8 land parcels in the same graph. However, for other scenarios that have many more land parcels this would be more difficult; e.g., in scenario 4B there are 53 land parcels.

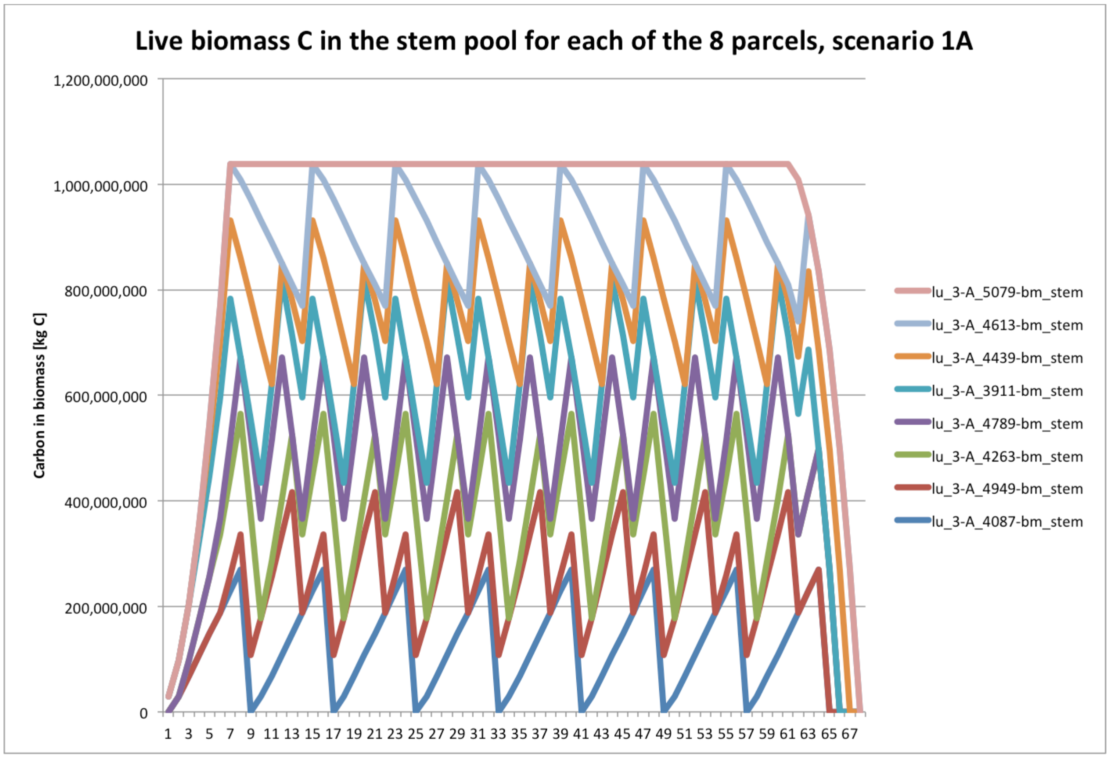

Figure C.2. Live biomass C in the stem pool for each of the 8 parcels, scenario 1A

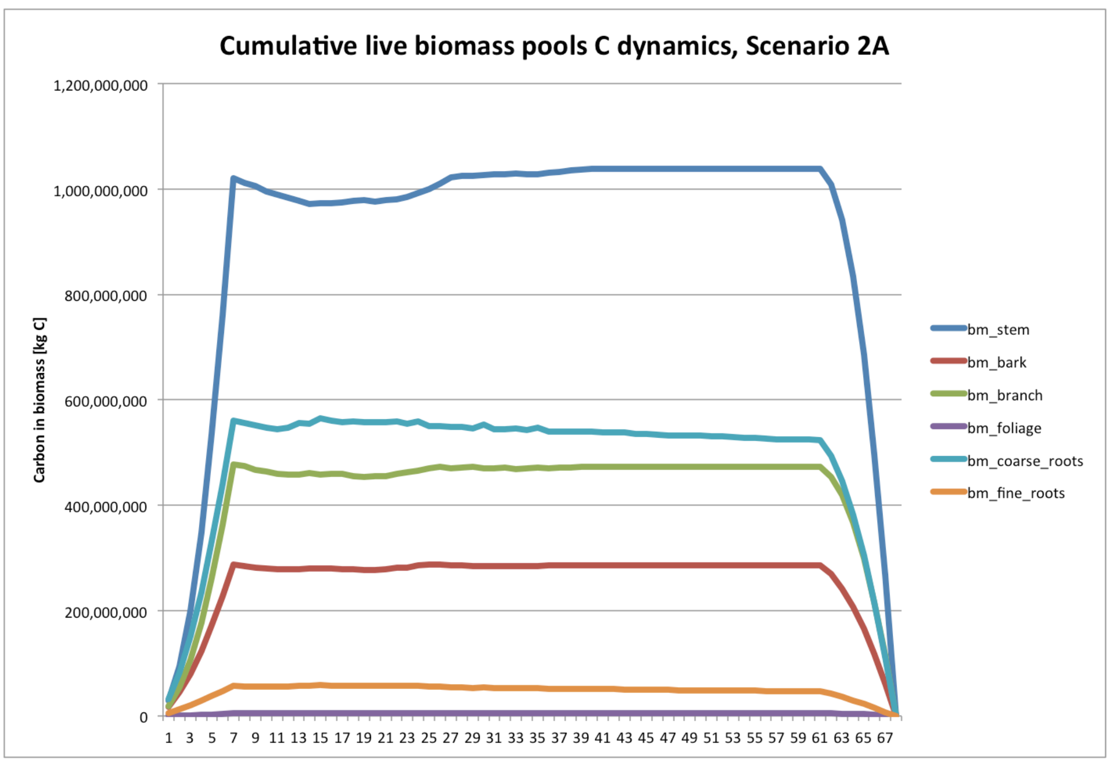

Figure C.3. Live biomass pools dynamics in all land parcels, scenario 2A

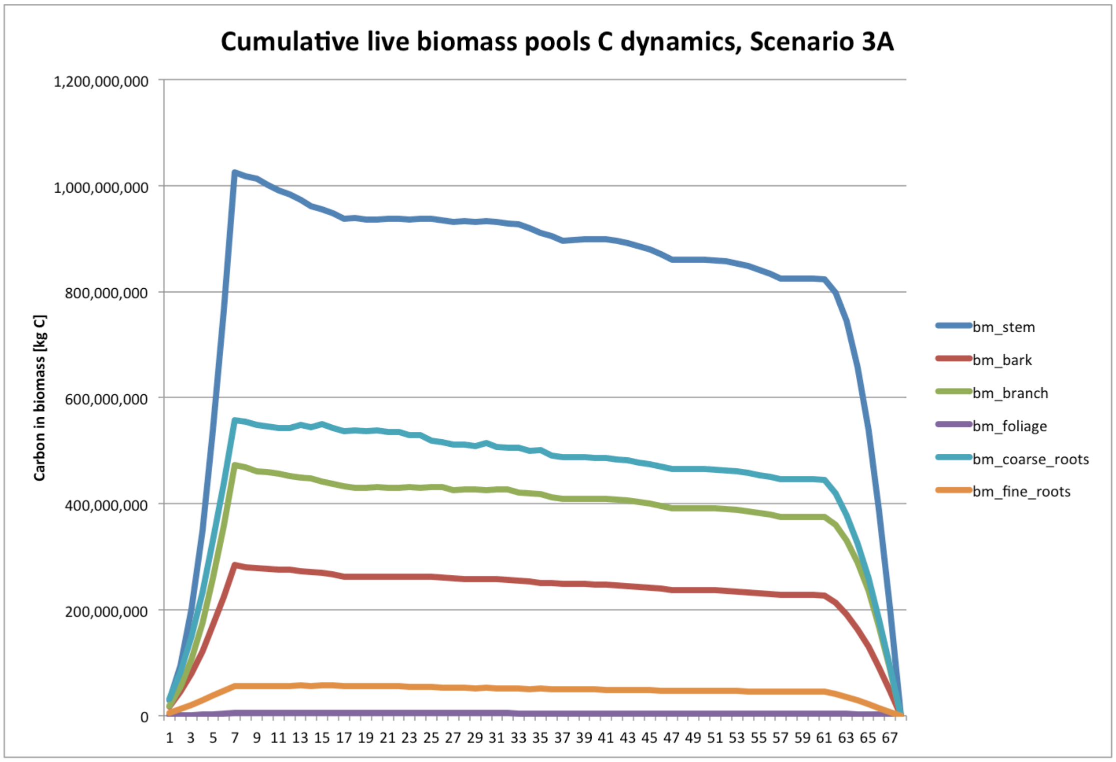

Figure C.4. Live biomass pools dynamics in all land parcels, scenario 3A

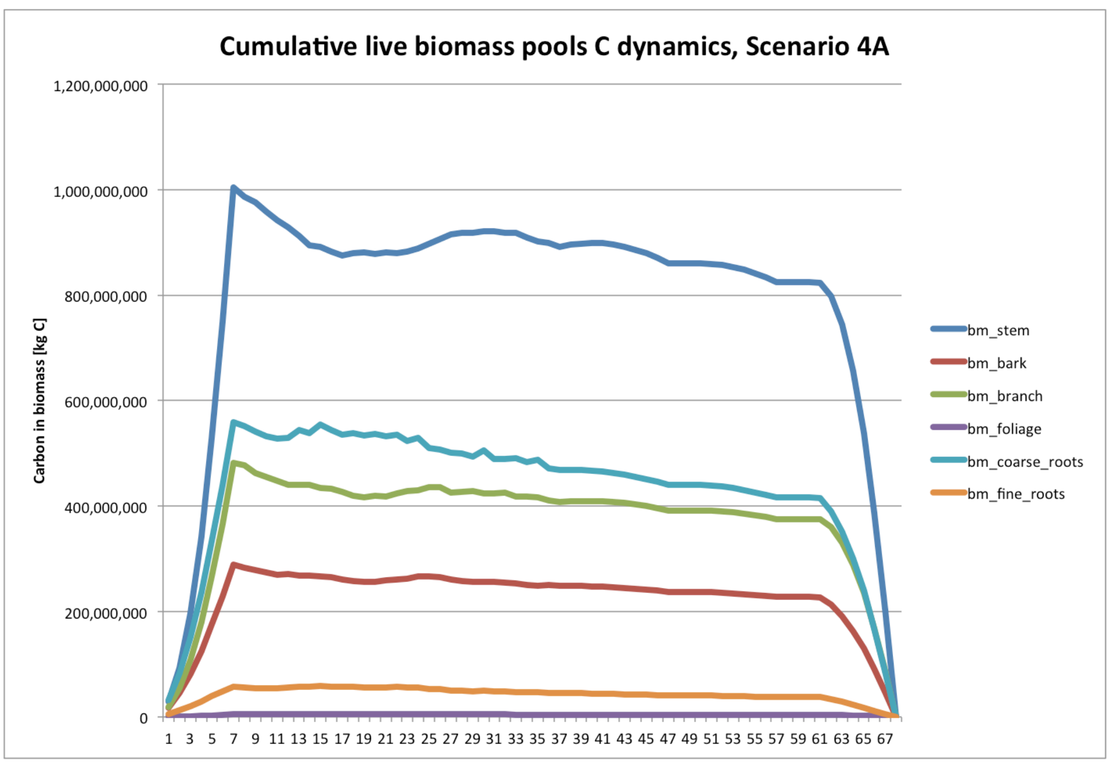

Figure C.5. Live biomass pools dynamics in all land parcels, scenario 4A

Scenarios 1B-4B did not allow idle treatments, therefore all harvested land parcels continued to be planted towards the end of the planning horizon instead of being abandoned, even though these parcels would not have the chance to be harvested again before the end of the scenario. This is equivalent to the plantation continuing to produce biomass for biofuel after the planning horizon.

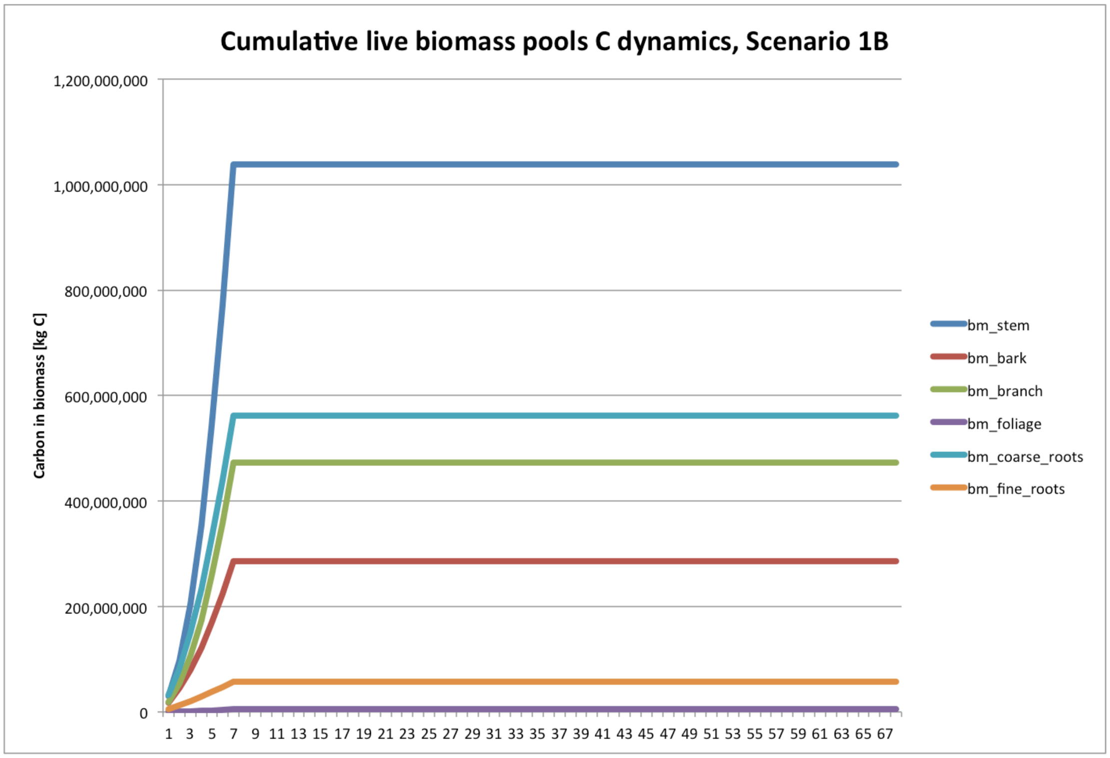

Figure C.6. Live biomass pools dynamics in all land parcels, scenario 1B

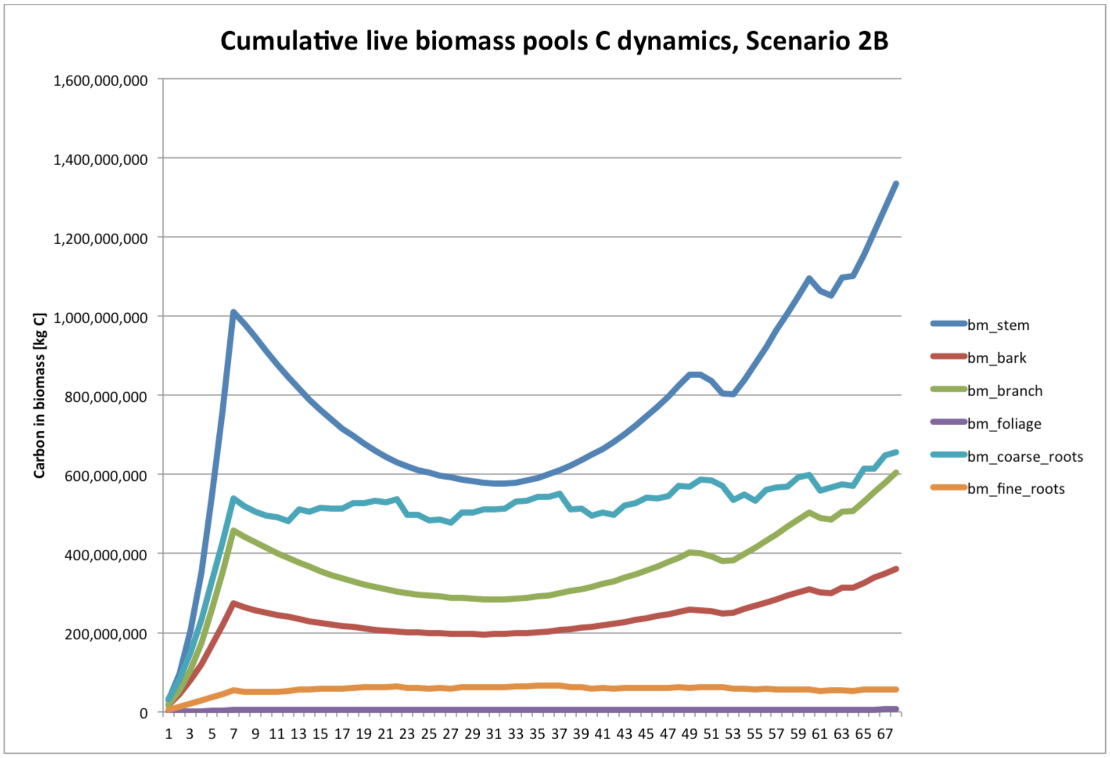

Figure C.7. Live biomass pools dynamics in all land parcels, scenario 2B

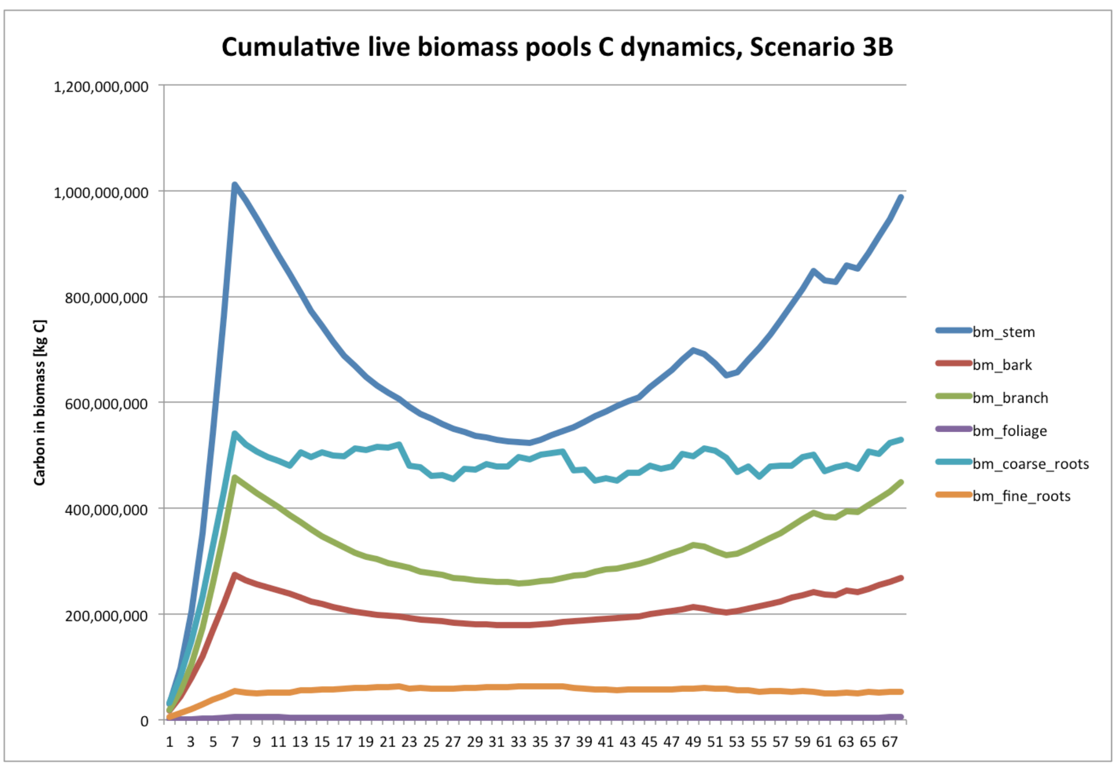

Figure C.8. Live biomass pools dynamics in all land parcels, scenario 3B

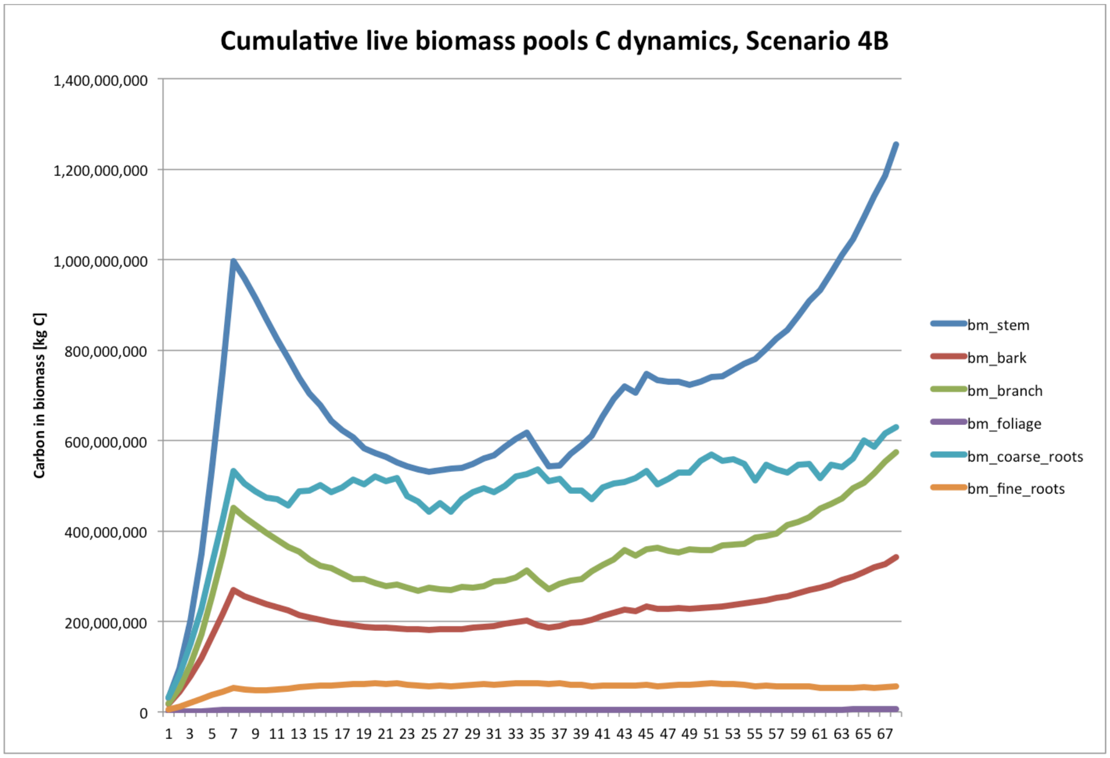

Figure C.9. Live biomass pools dynamics in all land parcels, scenario 4B

For illustrative purposes, we include below 3 examples of live biomass dynamics for the six pools by individual land parcels.

The first example is from scenario 1A, which produced a somewhat less complicated solution; the model solution generated harvesting-regeneration plans only for land unit LU_3, dividing it into eight equal-area land parcels, each with 11,129.89 hectares. The dynamics of the biomass pools for one of these land parcels are exhibited in Figure C.10, showing a sequence of eight treatments of type S, each with a rotation age of 8 years.

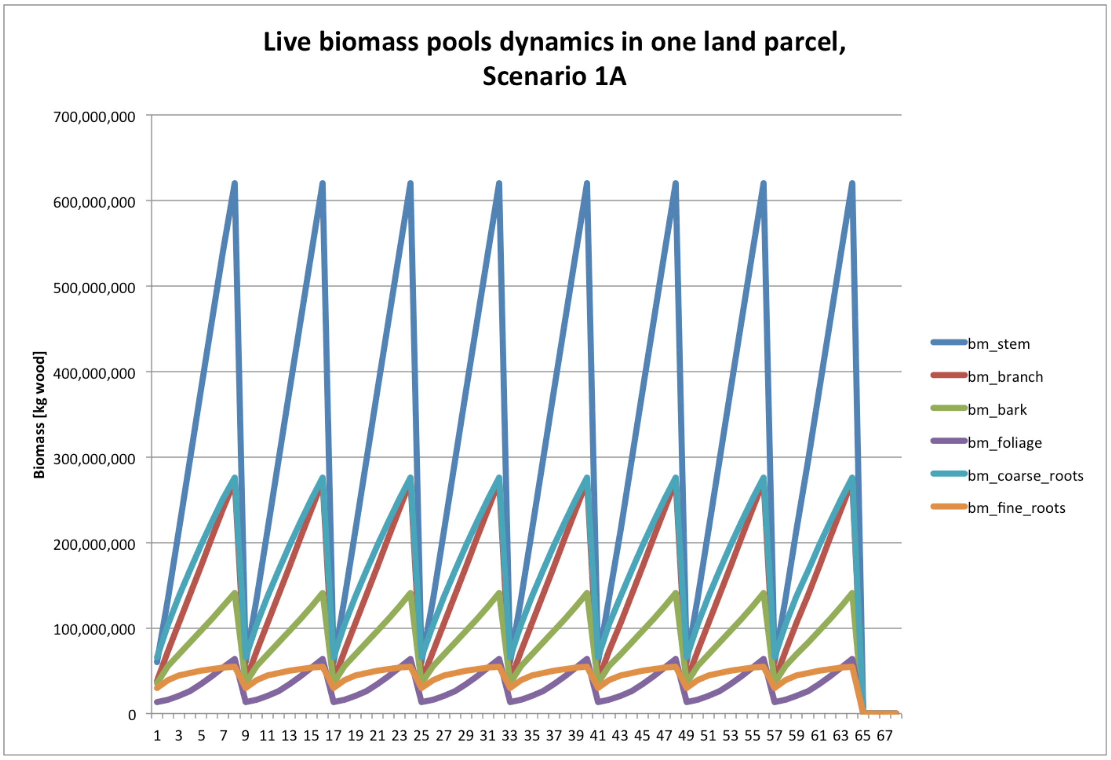

Figure C.10. Live biomass pools dynamics in one land parcel, scenario 1A

The second example is from scenario 2A, which produced a somewhat more complicated solution; the model solution generated harvesting-regeneration plans for land units LU_2, LU_3, and LU_5 and a total number of 58 land parcels. The dynamics of the biomass pools for one of these land parcels in LU_3 is shown in Figure C.11. The first treatment is S (8 years), followed by a treatment C (15 years, including two intermediate and one final harvests), and five more S treatments afterwards. In this figure the decadal increases in biomass yield are evident.

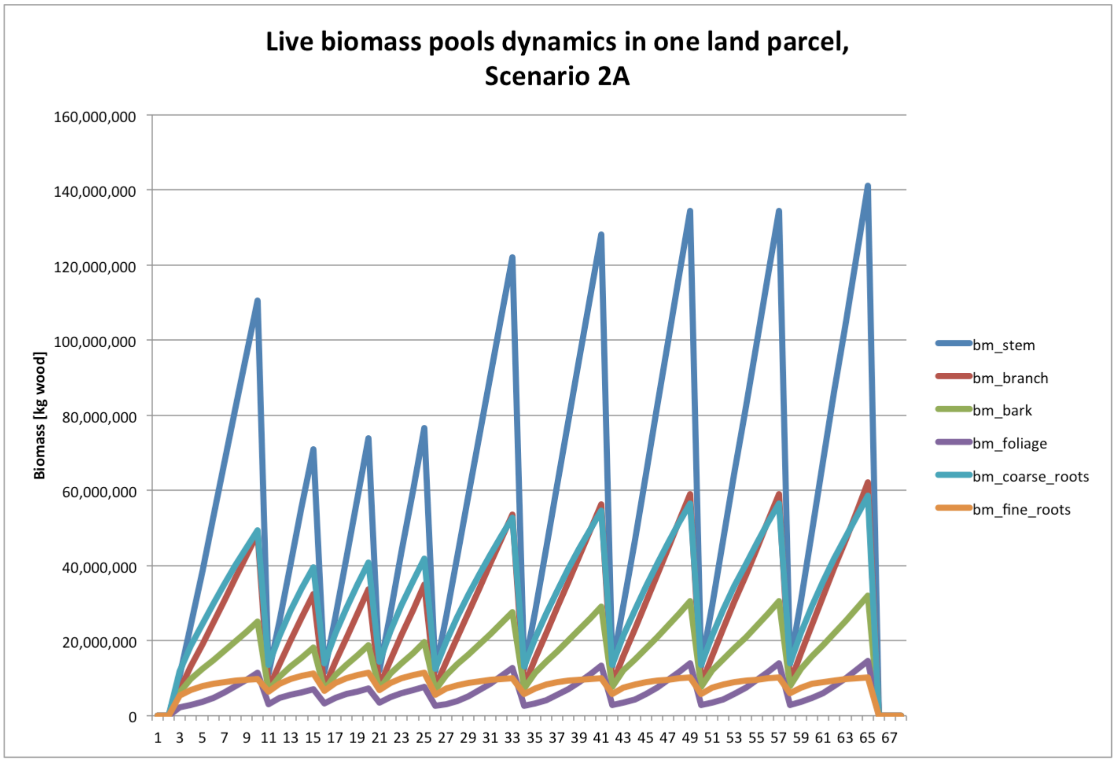

Figure C.11. Live biomass pools dynamics in one land parcel, scenario 2A

The third example is from scenario 4A, which produced a solution that included land parcels in an irrigated land unit, LU_2. The dynamics of the biomass pools for one of these land parcels in LU_2 is shown in Figure C.12. The first treatment is Si (6 years), followed by a sequence of treatments Ci (15 years each, including four intermediate and one final harvests). The decadal increases in biomass yield are evident (from one treatment to the next), as well as the slight increases within the same rotation (within the same treatment, between the intermediate harvests), as expected.

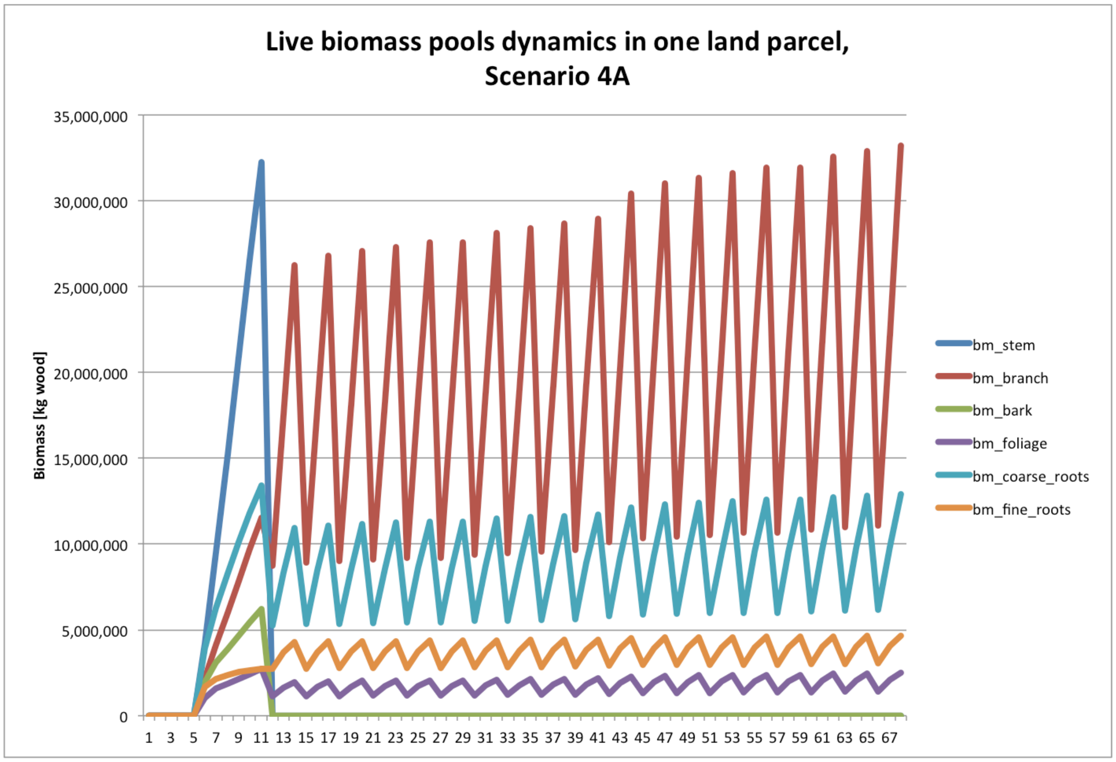

Figure C.12. Live biomass pools dynamics in one land parcel, scenario 2A

APPENDIX D. Detailed scenario results

Table D.1. C-BOS model results by land unit for scenario 1A: no improvements in biomass yield or conversion efficiency; idle treatment allowed

Land unit type	Max annual area [ha/yr]	Net average area [ha/yr]	Site prep. unit cost [$/t]	Land rent unit cost [$/t]	Biomass production unit cost [$/t]	Biomass transport unit cost [$/t]	Harvested biomass (stem+bark +branch) [t]
LU_1	0	0	0	0	0	0	0
LU_2	0	0	0	0	0	0	0
LU_3	89,039	79,873	0.30	14.52	81.25	42.89	59,167,192
LU_4	0	0	0	0	0	0	0
LU_5	0	0	0	0	0	0	0
LU_6	0	0	0	0	0	0	0

Table D.2. C-BOS model results by land unit for scenario 2A: improvements in biomass yield; idle treatment allowed

Land unit type	Max annual area [ha/yr]	Net average area [ha/yr]	Site prep. unit cost [$/t]	Land rent unit cost [$/t]	Biomass production unit cost [$/t]	Biomass transport unit cost [$/t]	Harvested biomass (stem+bark +branch) [t]
LU_1	0	0	0	0	0	0	0
LU_2	159	70	0	11.58	93.21	45.97	81,396
LU_3	79,747	68,598	0.28	12.81	76.74	42.89	57,592,567
LU_4	0	0	0	0	0	0	0
LU_5	7,747	2,410	0.99	12.47	94.38	28.67	1,559,177
LU_6	0	0	0	0	0	0	0

Table D.3. C-BOS model results by land unit for scenario 3A: improvements in conversion efficiency; idle treatment allowed

Land unit type	Max annual area [ha/yr]	Net average area [ha/yr]	Site prep. unit cost [$/t]	Land rent unit cost [$/t]	Biomass production unit cost [$/t]	Biomass transport unit cost [$/t]	Harvested biomass (stem+bark +branch) [t]
LU_1	0	0	0	0	0	0	0
LU_2	169	74	0	12.17	97.20	45.97	82,246
LU_3	84,238	70,421	0.32	14.50	81.46	42.89	52,240,608
LU_4	0	0	0	0	0	0	0
LU_5	2,392	501	1.52	12.83	95.71	28.67	314,934
LU_6	0	0	0	0	0	0	0

Table D.4. C-BOS model results by land unit scenario 4A: improvements in both biomass yield and conversion efficiency; idle treatment allowed

Land unit type	Max annual area [ha/yr]	Net average area [ha/yr]	Site prep. unit cost [$/t]	Land rent unit cost [$/t]	Biomass production unit cost [$/t]	Biomass transport unit cost [$/t]	Harvested biomass (stem+bark +branch) [t]
LU_1	0	0	0	0	0	0	0
LU_2	325	144	0	11.58	93.21	45.97	166,670
LU_3	71,443	59,288	0.29	12.84	77.11	42.89	49,647,185
LU_4	0	0	0	0	0	0	0
LU_5	14,612	4,515	1.00	12.47	94.41	28.67	2,919,349
LU_6	0	0	0	0	0	0	0

Table D.5. C-BOS model results by land unit for scenario 1B: no improvements in biomass yield or conversion efficiency; no idle treatment

Land unit type	Max annual area [ha/yr]	Net average area [ha/yr]	Site prep. unit cost [$/t]	Land rent unit cost [$/t]	Biomass production unit cost [$/t]	Biomass transport unit cost [$/t]	Harvested biomass (stem+bark +branch) [t]
LU_1	0	0	0	0	0	0	0
LU_2	49	45	0	11.13	92.65	46.95	54,013
LU_3	79,277	75,562	0.26	13.37	82.56	42.87	60,779,457
LU_4	0	0	0	0	0	0	0
LU_5	0	0	0	0	0	0	0
LU_6	0	0	0	0	0	0	0

Table D.6. C-BOS model results by land unit for scenario 2B: improvements in biomass yield; no idle treatment

Land unit type	Max annual area [ha/yr]	Net average area [ha/yr]	Site prep. unit cost [$/t]	Land rent unit cost [$/t]	Biomass production unit cost [$/t]	Biomass transport unit cost [$/t]	Harvested biomass (stem+bark +branch) [t]
LU_1	0	0	0	0	0	0	0
LU_2	159	70	0	11.58	93.21	45.97	81,396
LU_3	79,747	68,598	0.28	12.81	76.74	42.89	57,592,567
LU_4	0	0	0	0	0	0	0
LU_5	7,747	2,410	0.99	12.47	94.38	28.67	1,559,177
LU_6	0	0	0	0	0	0	0

Table D.7. C-BOS model results by land unit for scenario 3B: improvements in conversion efficiency; no idle treatment

Land unit type	Max annual area [ha/yr]	Net average area [ha/yr]	Site prep. unit cost [$/t]	Land rent unit cost [$/t]	Biomass production unit cost [$/t]	Biomass transport unit cost [$/t]	Harvested biomass (stem+bark +branch) [t]
LU_1	0	0	0	0	0	0	0
LU_2	108	103	0	12.58	101.66	47.03	109,848
LU_3	79,204	75,490	0.29	15.02	87.76	42.87	54,038,522
LU_4	0	0	0	0	0	0	0
LU_5	0	0	0	0	0	0	0
LU_6	0	0	0	0	0	0	0

Table D.8. C-BOS model results by land unit scenario 4B: improvements in both biomass yield and conversion efficiency; no idle treatment

Land unit type	Max annual area [ha/yr]	Net average area [ha/yr]	Site prep. unit cost [$/t]	Land rent unit cost [$/t]	Biomass production unit cost [$/t]	Biomass transport unit cost [$/t]	Harvested biomass (stem+bark +branch) [t]
LU_1	0	0	0	0	0	0	0
LU_2	948	882	0	11.08	92.91	47.17	1,070,109
LU_3	75,154	71,753	0.26	13.34	82.87	42.87	57,828,727
LU_4	0	0	0	0	0	0	0
LU_5	0	0	0	0	0	0	0
LU_6	0	0	0	0	0	0	0

Figure D.1 through Figure D.8 show the total above-ground biomass harvested in each of the eight scenarios. As expected, in the non-improvement scenarios 1A and 1B the harvested biomass volume is constant throughout the planning horizon, starting with year 8 – the first harvest year.

In scenario 2A the harvested volume is fairly constant throughout the years, since the quantity of ethanol needed (and therefore of biomass) is constant. In scenarios 3A and 4A the volume of harvested biomass is decreasing throughout the decades, because less and less biomass is needed due to the increased conversion efficiency (Figure D.3 and Figure D.4).

In Figure D.3 and Figure D.4 the decadal improvement steps in conversion efficiency are represented by corresponding reduction steps in the biomass harvested.

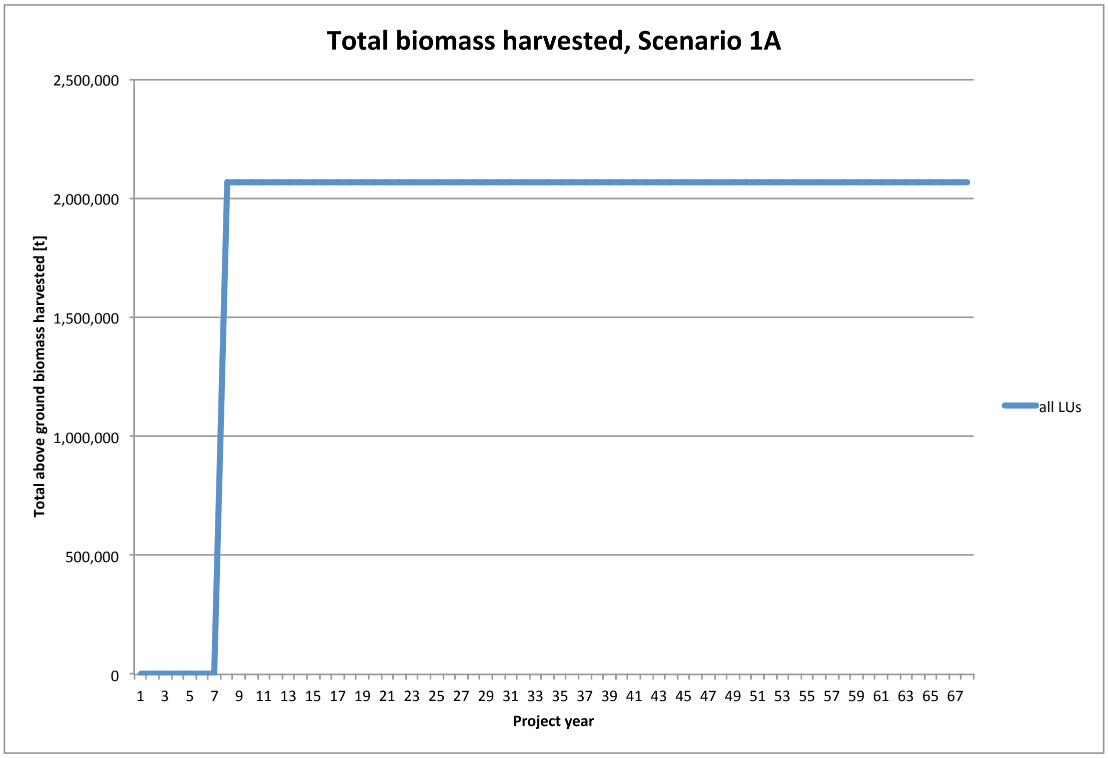

Figure D.1. Total above-ground biomass harvested in all land units, scenario 1A

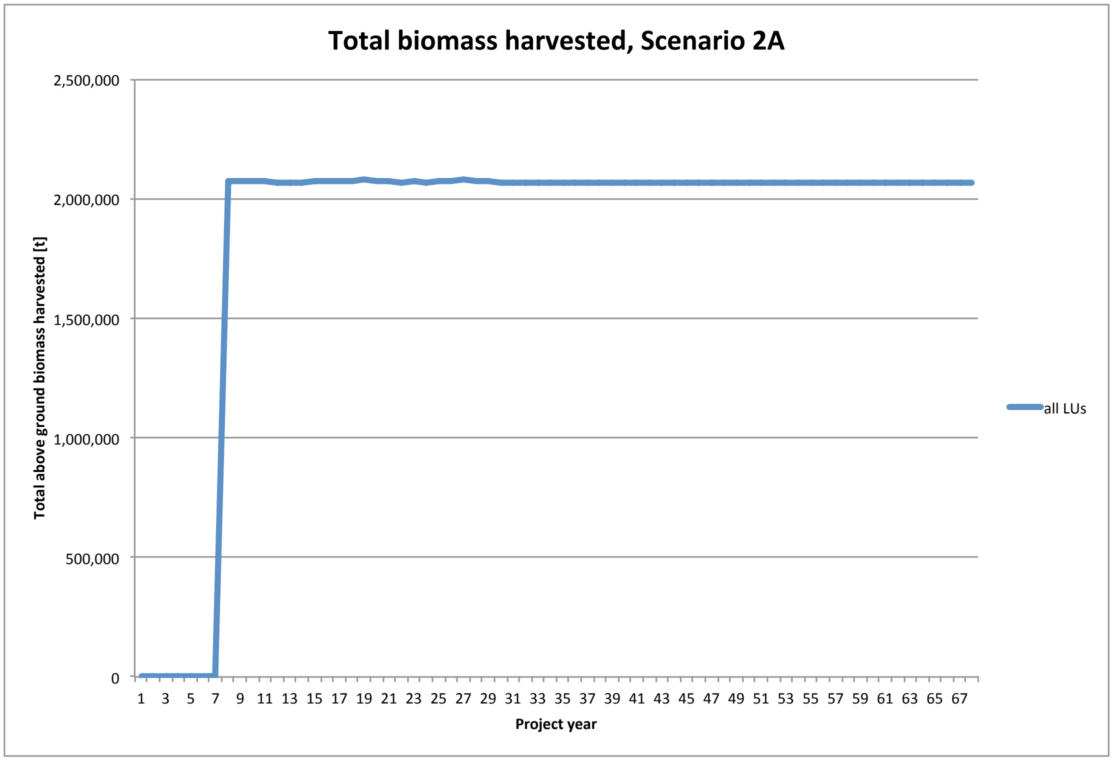

Figure D.2. Total above-ground biomass harvested in all land units, scenario 2A

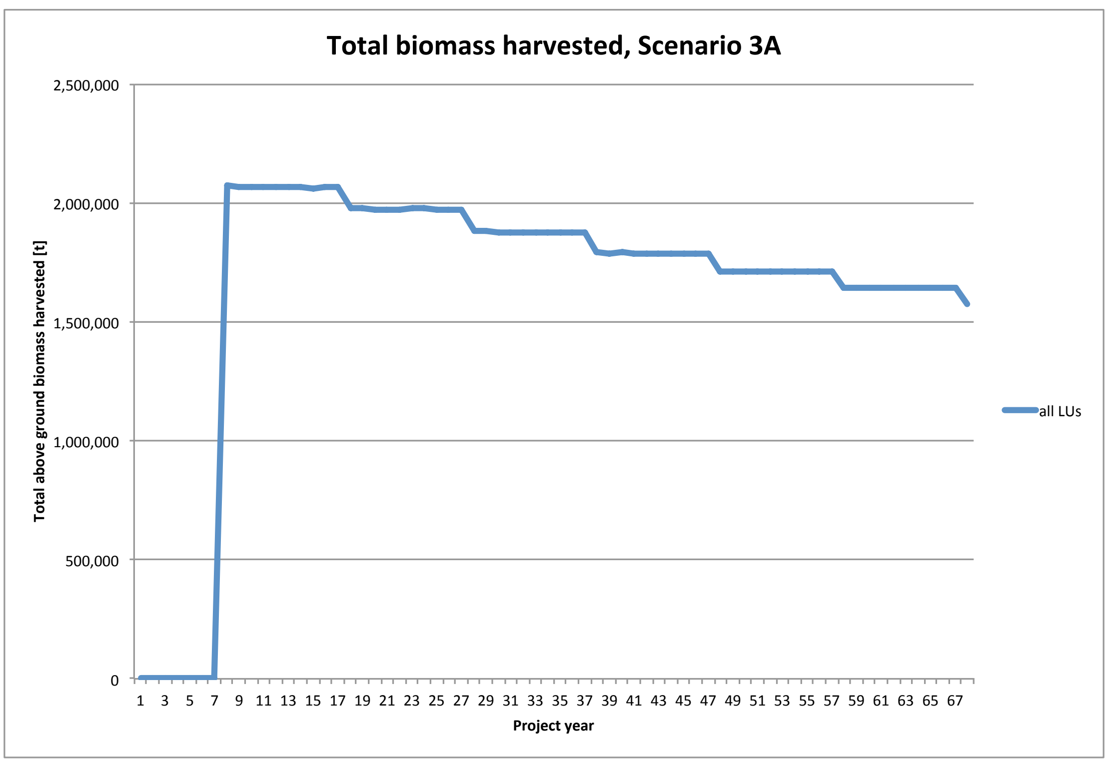

Figure D.3. Total above-ground biomass harvested in all land units, scenario 3A

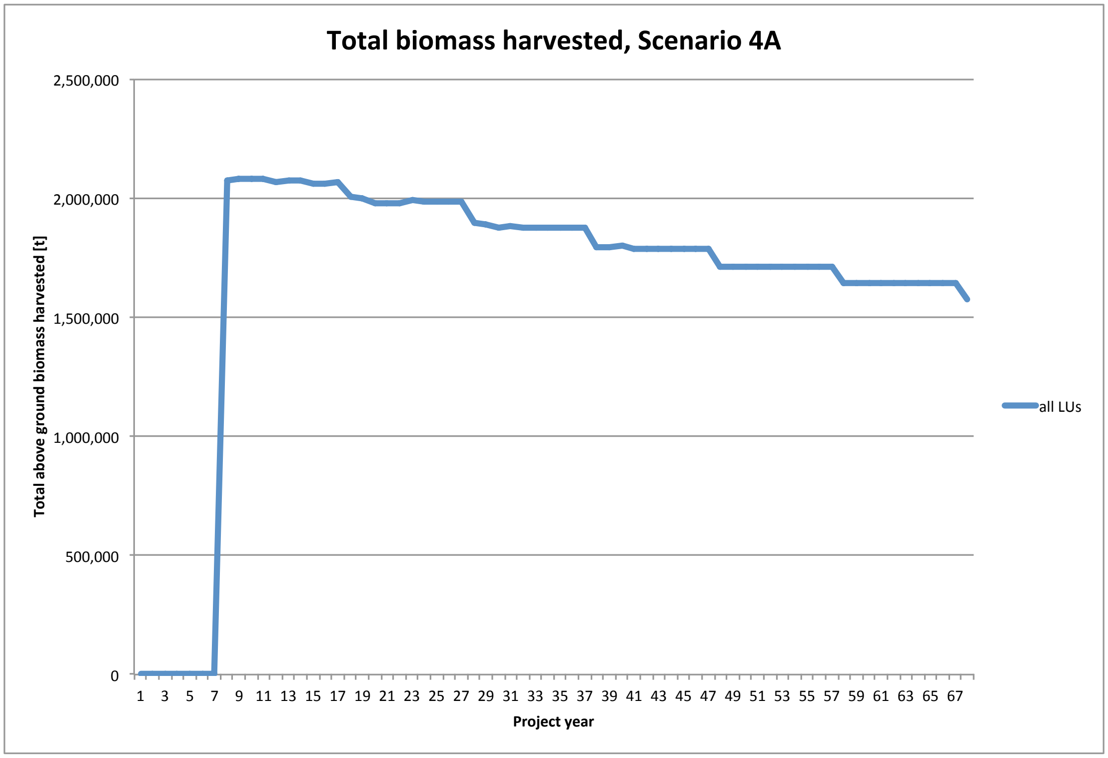

Figure D.4. Total above-ground biomass harvested in all land units, scenario 4A

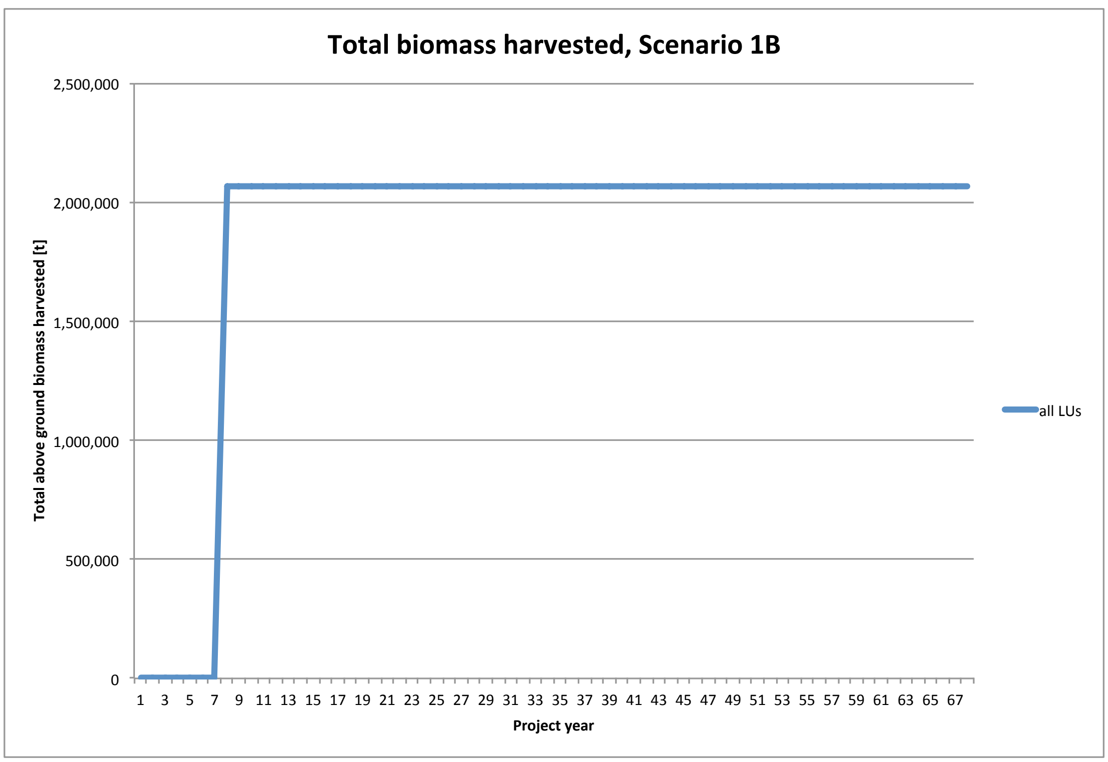

Figure D.5. Total above-ground biomass harvested in all land units, scenario 1B

The improvements in biomass yield of scenario 2B result in the additional biomass harvested in the last portion of the planning horizon, as shown in Figure D.6. This biomass must be grown and harvested, because the hectares cannot be left empty or unused: the idle treatment is not available in this scenario, and therefore the model must use one of the available production treatments (*i.e.* S, Si, C, or Ci).

The improvements in conversion efficiency of scenario 3B, which require less biomass to be harvested, do show as a downward trend in Figure D.7. However, towards the end of the planning horizon some biomass needs to be planted and harvested, resulting in the spikes of additional biomass.

The general downward trend of harvested biomass did not manifest in scenario 4B (Figure D.8). Besides the conversion efficiency improvements considered in scenario 3B, in scenario 4B the biomass yield improvements were also considered. Since the hectares could not be left empty (*i.e.* one of the production treatments had to be used), the biomass yield improvements generated a surplus of biomass volume, which counterbalanced the reduction in biomass due to the improvements in conversion efficiency.

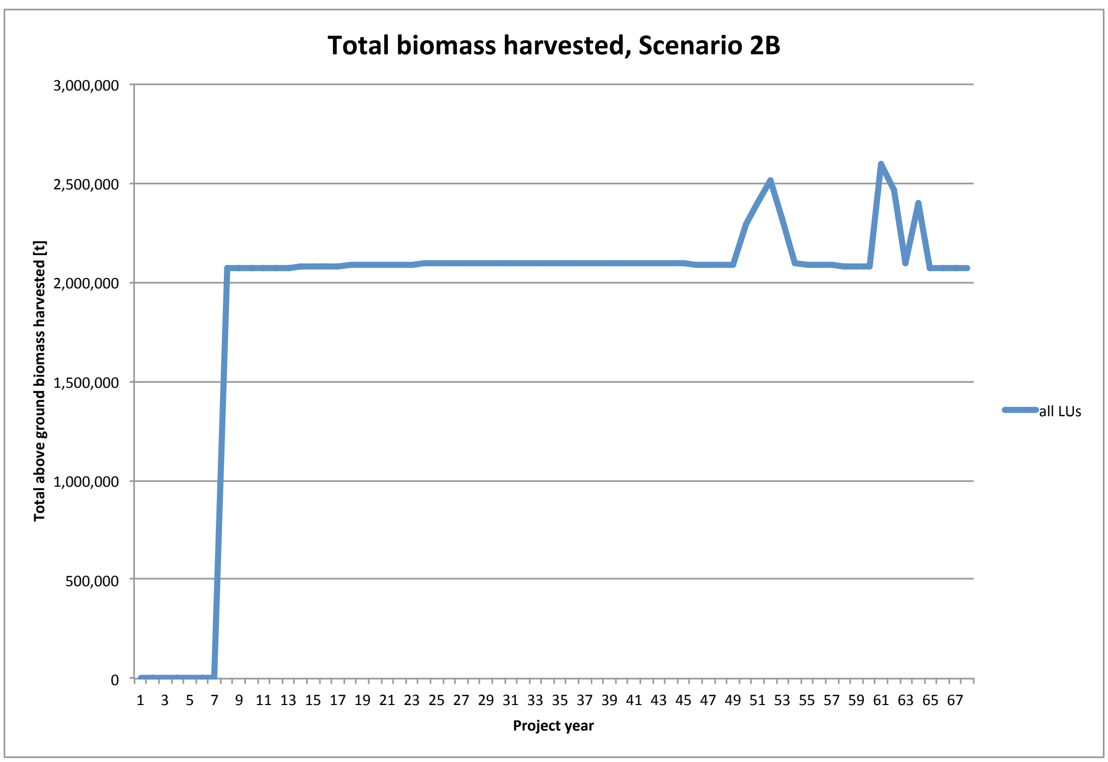

Figure D.6. Total above-ground biomass harvested in all land units, scenario 2B

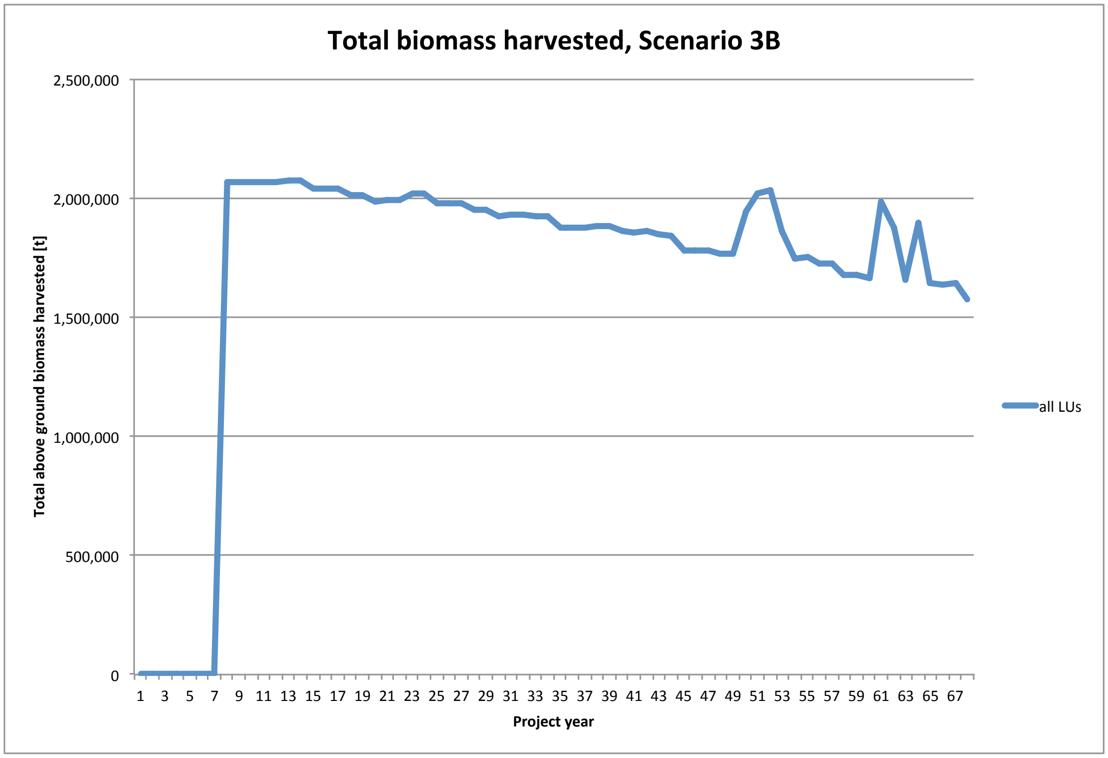

Figure D.7. Total above-ground biomass harvested in all land units, scenario 3B

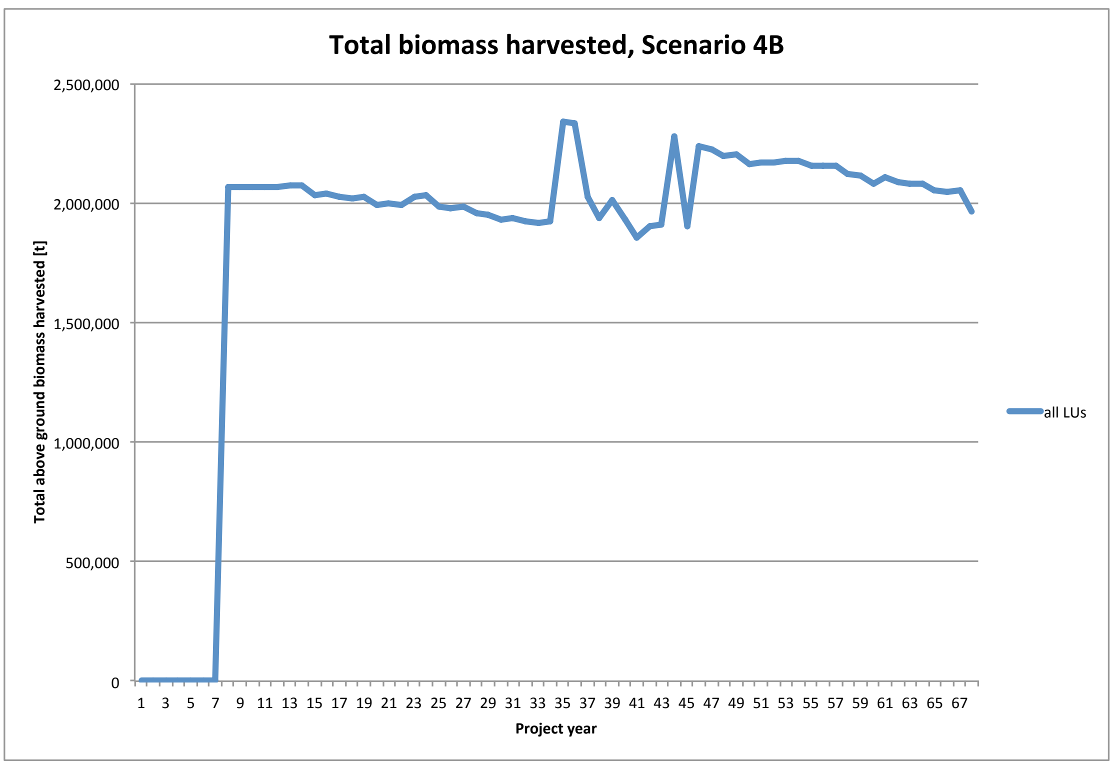

Figure D.8. Total above-ground biomass harvested in all land units, scenario 4B

APPENDIX E. Life cycle carbon dynamics in live biomass and dead organic matter pools

Life cycle carbon dynamics in live biomass pools

The life cycle carbon dynamics in live biomass pools for all scenarios are shown in Figure E.1 through Figure E.4. The life cycle carbon dynamics of scenario 1 are identical to those of scenario 5, since the biomass yield, the conversion efficiency, and the ethanol demand are the same for the two scenarios, as well as the presence of the idle treatment (Figure E.1). Similarly, the life cycle carbon dynamics of scenarios 2 and 6, 3 and 7, and 4 and 8 are the same.

In scenarios 1 and 5 (Figure E.1) the biomass (and its carbon content, shown in the graph) is accumulating rapidly in all pools (except for foliage which is grown and lost each year) during the first 8 years of the project, after which ethanol production (hence, harvest) begins, and the biomass volumes remain constant until the end of the project because of the constant minimum demand of ethanol volume.

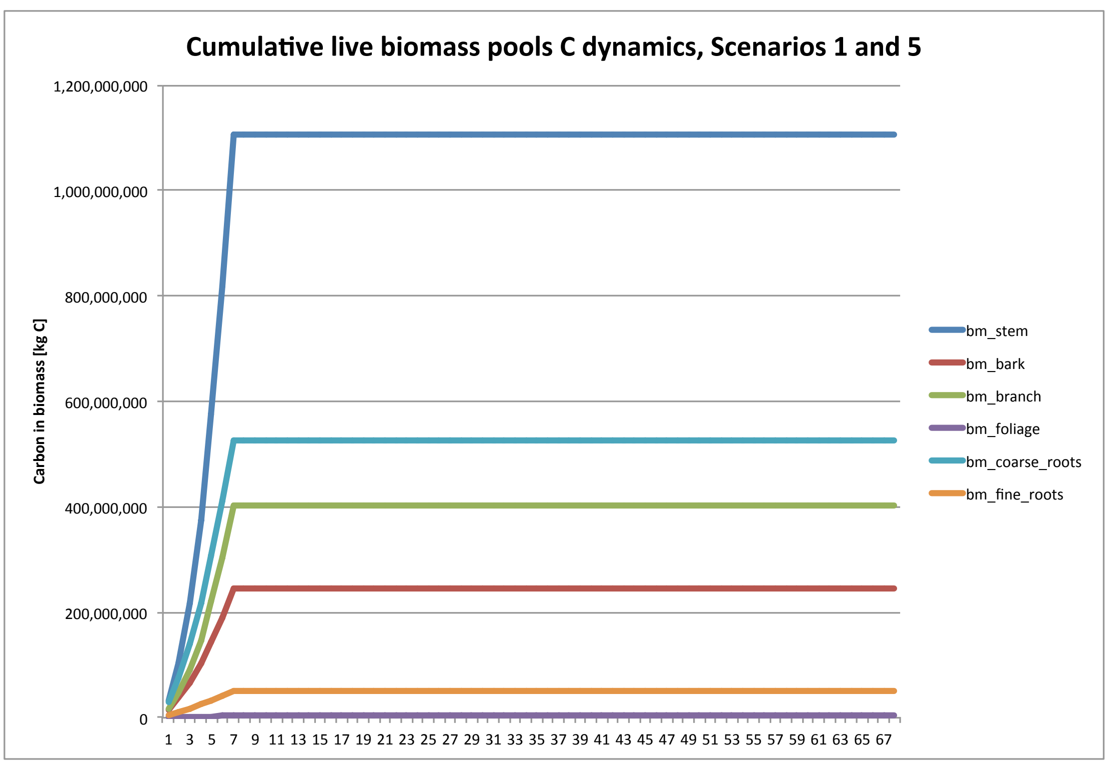

Figure E.1. Life cycle live biomass carbon dynamics, scenarios 1 and 5

In scenarios 2 and 6 (Figure E.2) the carbon dynamic is similar with scenarios 1 and 5, except for the last years of the project when the idle treatments allow plantation hectares to be abandoned because less and less hectares need to be planted, as remaining biomass is harvested just enough to provide the feedstock for the ethanol needed. As there is no incentive in the model to incur costs for planting hectares that will not be harvested during the planning horizon, in the final year all remaining biomass is harvested, emptying the live biomass carbon pools completely.

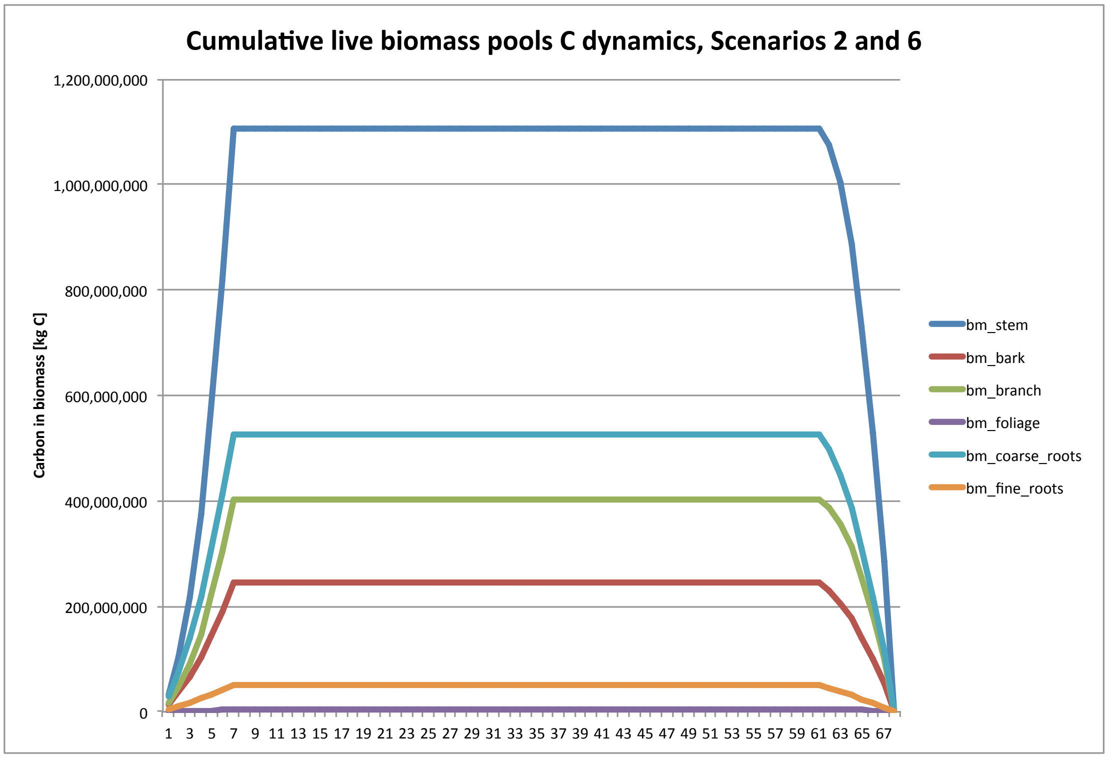

Figure E.2. Life cycle live biomass carbon dynamics, scenarios 2 and 6

The live biomass carbon dynamics in scenarios 3 and 7 (Figure E.3) illustrate how much variation can there be from year to year throughout the planning horizon. The capability of the Bio-CarbD model to account for these year-to-year changes allows for a more informative insight in the carbon dynamics and carbon balance analysis.

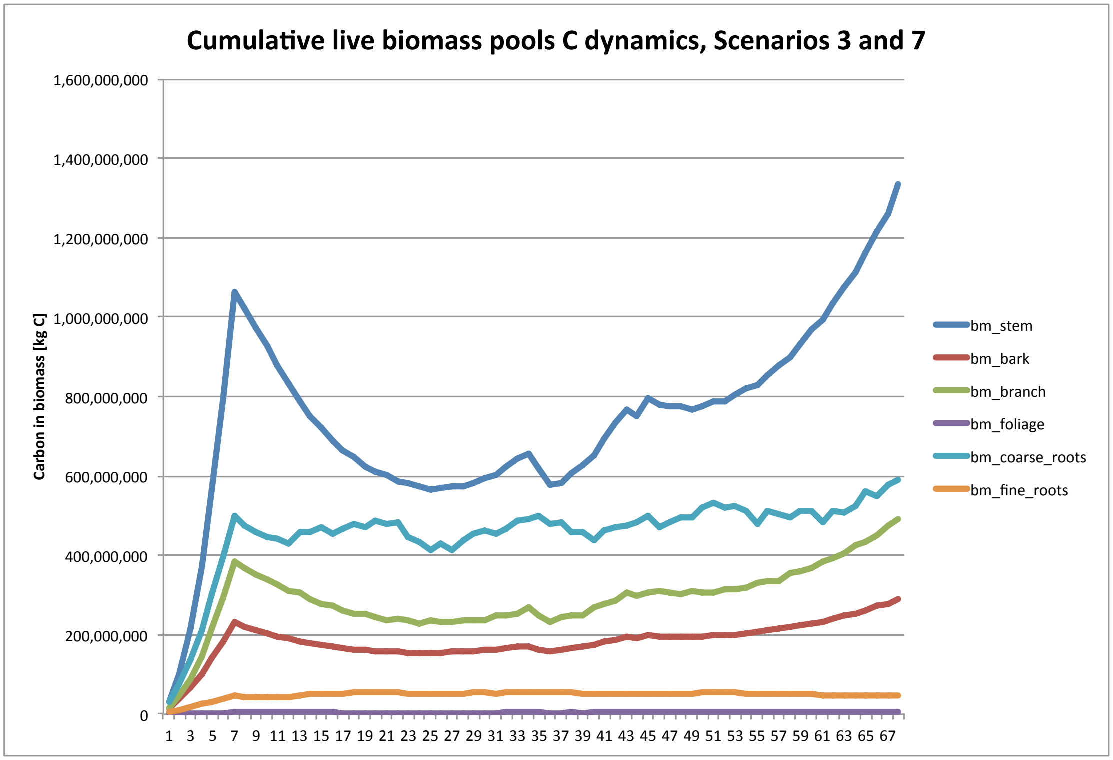

Figure E.3. Life cycle live biomass carbon dynamics, scenarios 3 and 7

For the live biomass carbon dynamics in scenarios 4 and 8 there are three distinct regions: 1) in the first 8 years biomass is accumulating fast; 2) from year 8 to about 60 the quantity of carbon in live biomass decreases slowly throughout the decades, as a consequence of the improved biomass yield and conversion efficiency and the idle treatments (*i.e.* less and less hectares need to be planted to produce the quantity of feedstock necessary for ethanol production); and, 3) after year 60 or so the plantation hectares are abandoned, no biomass needs to be planted anymore since there is no incentive to incur the costs of producing biomass that will not be harvested by the end of the planning horizon.

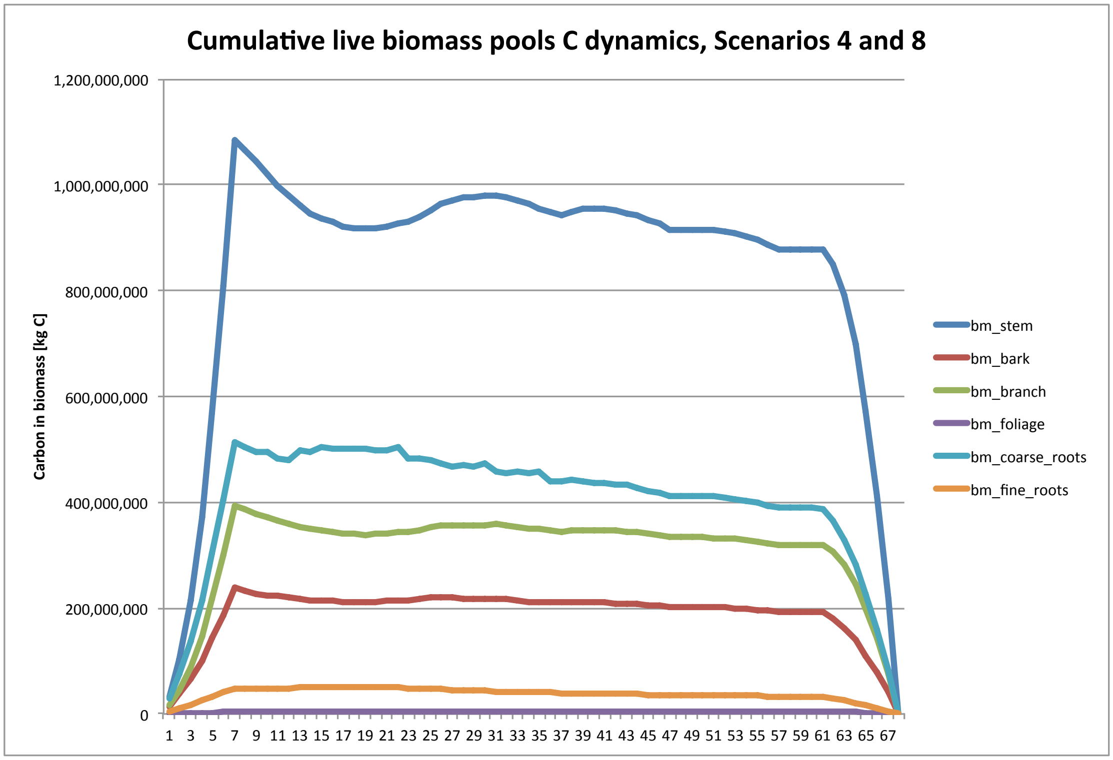

Figure E.4. Life cycle live biomass carbon dynamics, scenarios 4 and 8

Life cycle carbon dynamics in DOM pools

The life cycle carbon dynamics in dead organic matter pools for all scenarios are shown in Figure E. through Figure E.12. The slopes and shapes of the carbon dynamics curves are mainly determined by the decaying rate of the respective DOM pools.

Slow and medium decaying pools

In scenarios 1-4, where there are no initial C stocks in vegetation and soil, no losses from land-use change, and no losses from mechanized harvest-planting activities, the carbon is continually accumulating in the DOM slow decaying pools, both above- and below-ground (*i.e. dom_ag_slow* and *dom_bg_slow*), as well as in the *dom_medium* pool. This is expected since the rate of biomass accumulation in these pools is higher than their rate of decay, as described in the carbon transfer matrices in section 3.3.2. In scenario 2 less biomass accumulates in the slow decaying pools over the life of the project than scenario 1, since the area used is diminishing towards the end of the project for the former.

In scenarios 5-8, the below-ground slow decaying pools start with a certain quantity of carbon from the initial carbon stocks in DOM pools. As the carbon emissions from decay are proportional to the carbon existing in the DOM pools (as per the carbon transfer coefficients from DOM pools to the *CO2_dom* pool shown in Table 3.3 through Table 3.5), this results in more tonnes of carbon to be released per hectare. As a result, these emissions counterbalance the carbon accumulation in the slow decaying DOM pools, resulting in a small or negligible carbon sequestration in these pools. The DOM carbon losses from the impact of mechanized activities in harvest years and from natural decay factors also contribute to the low rate of carbon accumulation in these pools over the planning horizon, albeit by a smaller amount.

Fast decaying pools

The carbon accumulates continually in the fast decaying pools for the first few decades in scenarios 1-4, after which the rate of accumulation decreases. In the last years of scenario 2, the increase in accumulation can be attributed to the use of the idle treatment.

During the first few years of scenarios 5-8 the amount of biomass transferred from live biomass pools to DOM fast decaying pools is lower than the biomass that is lost through decay from these pools, so the net effect is that the carbon stock decreases. This is a direct consequence of the high carbon stock existing in the DOM pools before the project.

Very fast decaying pools

The carbon accumulated in the very fast decaying DOM pools (*dom_ag_very_fast* and *dom_bg_very_fast*) is fairly constant throughout the project in scenarios 5-8 (and after year 8 in scenarios 1-4), as expected, since the rate of turnover and decay is high, although carbon is continually transferred to these pools from the live biomass pools. The somewhat decreasing trend of the rate of accumulation in scenario 2 compared with scenario 1 can be justified similarly as for the fast decaying pools.

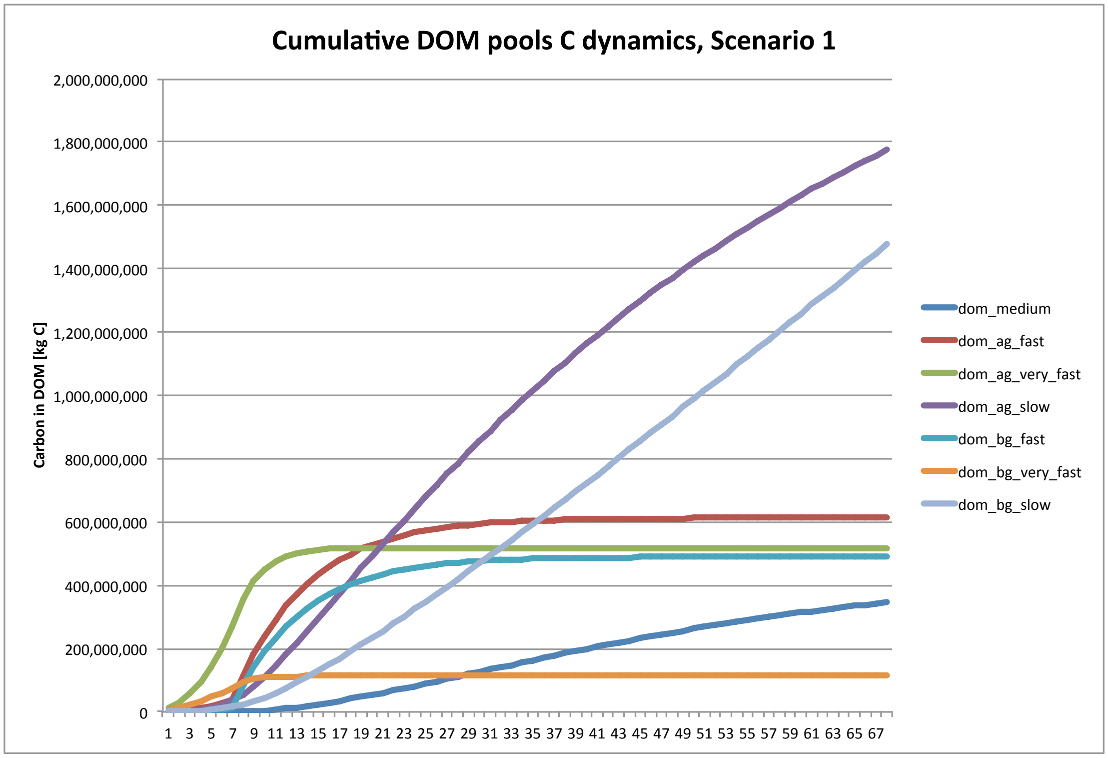

Figure E.5. Life cycle carbon dynamics in DOM pools, scenario 1

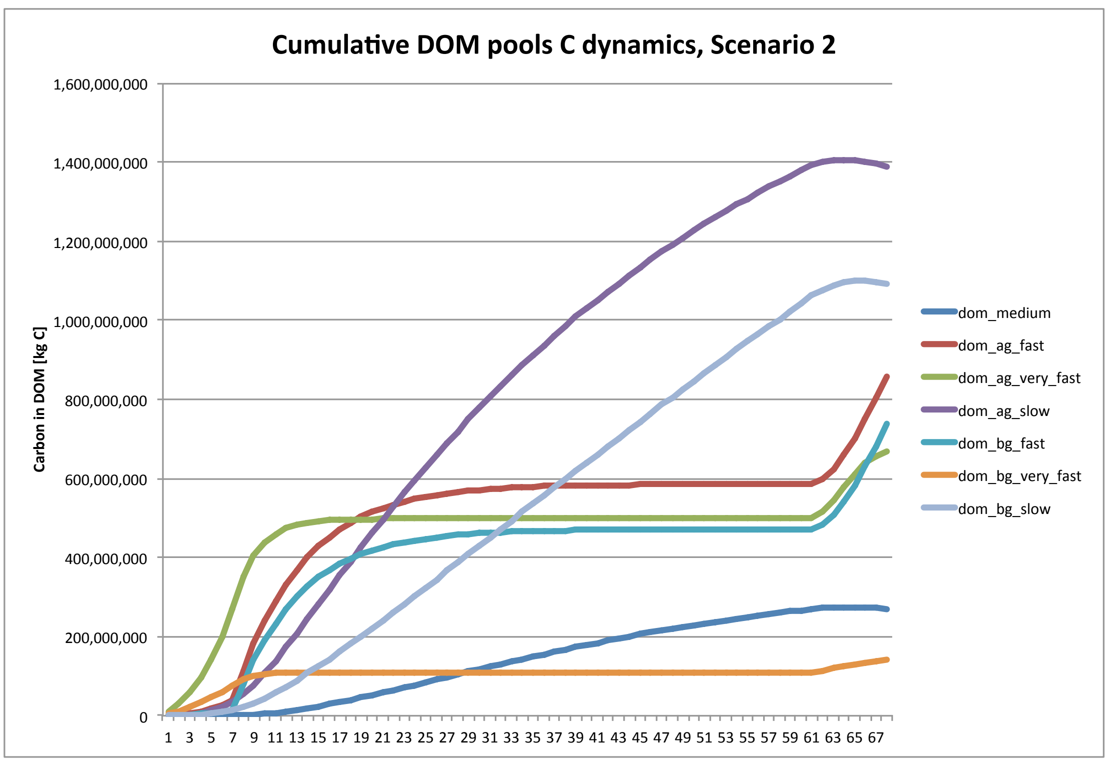

Figure E.6. Life cycle carbon dynamics in DOM pools, scenario 2

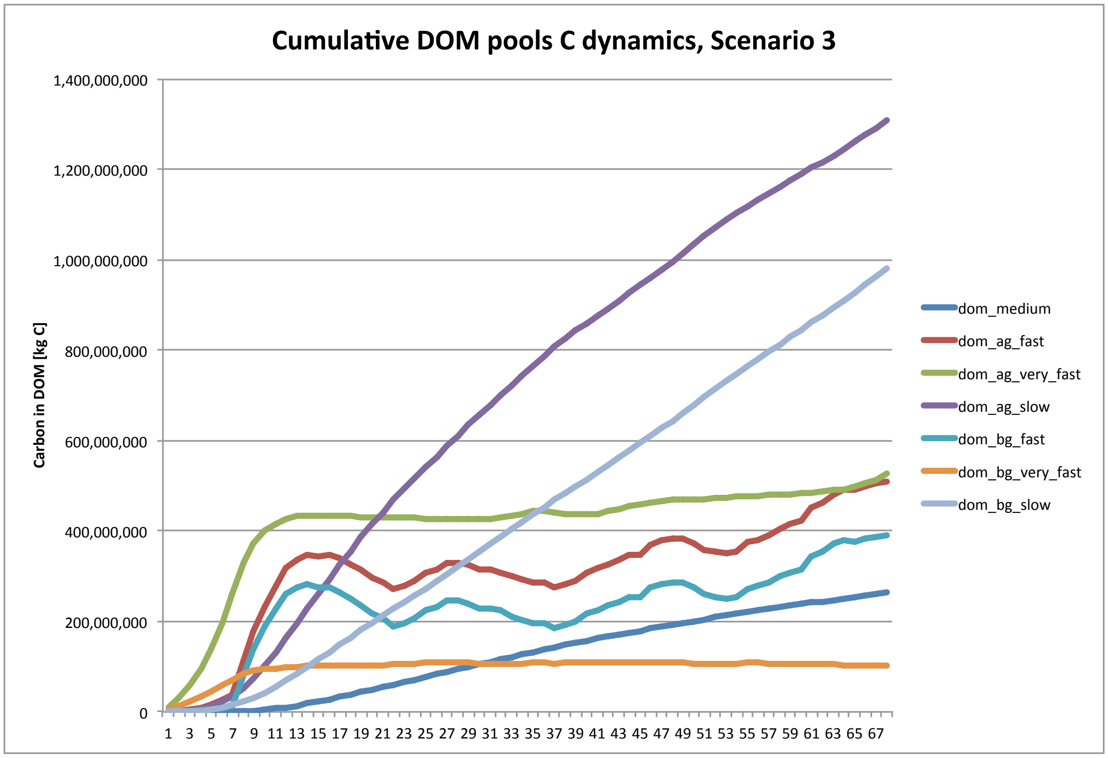

Figure E.7. Life cycle carbon dynamics in DOM pools, scenario 3

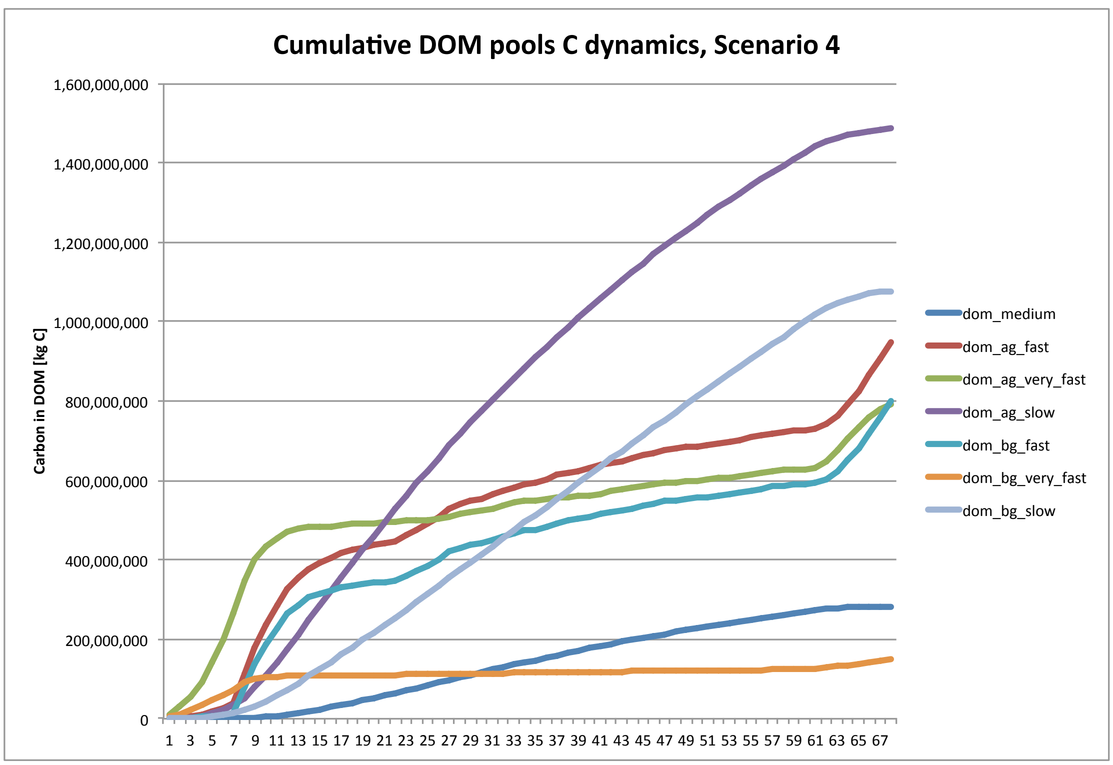

Figure E.8. Life cycle carbon dynamics in DOM pools, scenario 4

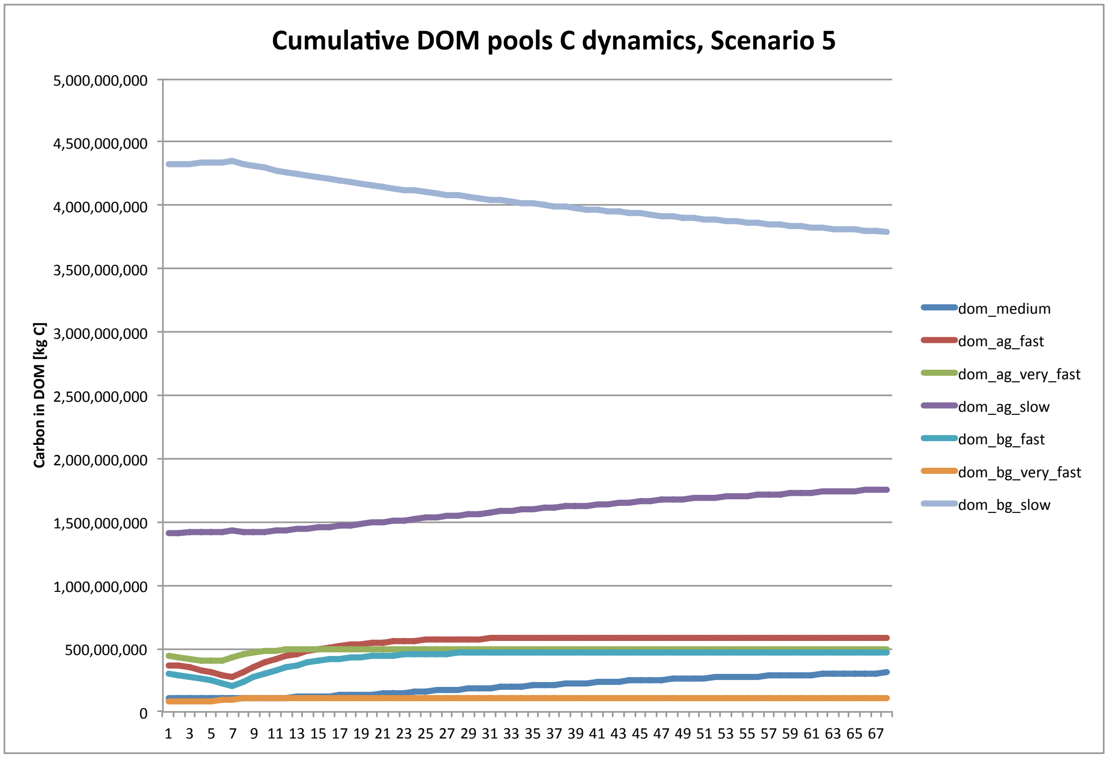

Figure E.9. Life cycle carbon dynamics in DOM pools, scenario 5

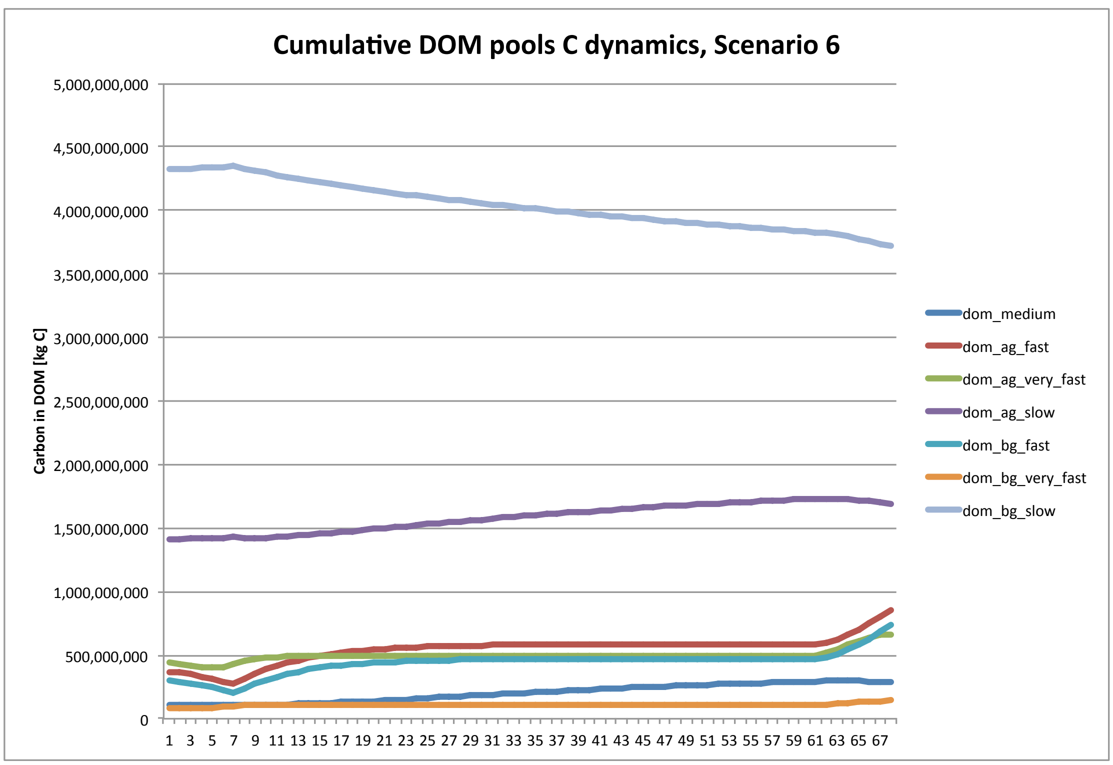

Figure E.10. Life cycle carbon dynamics in DOM pools, scenario 6

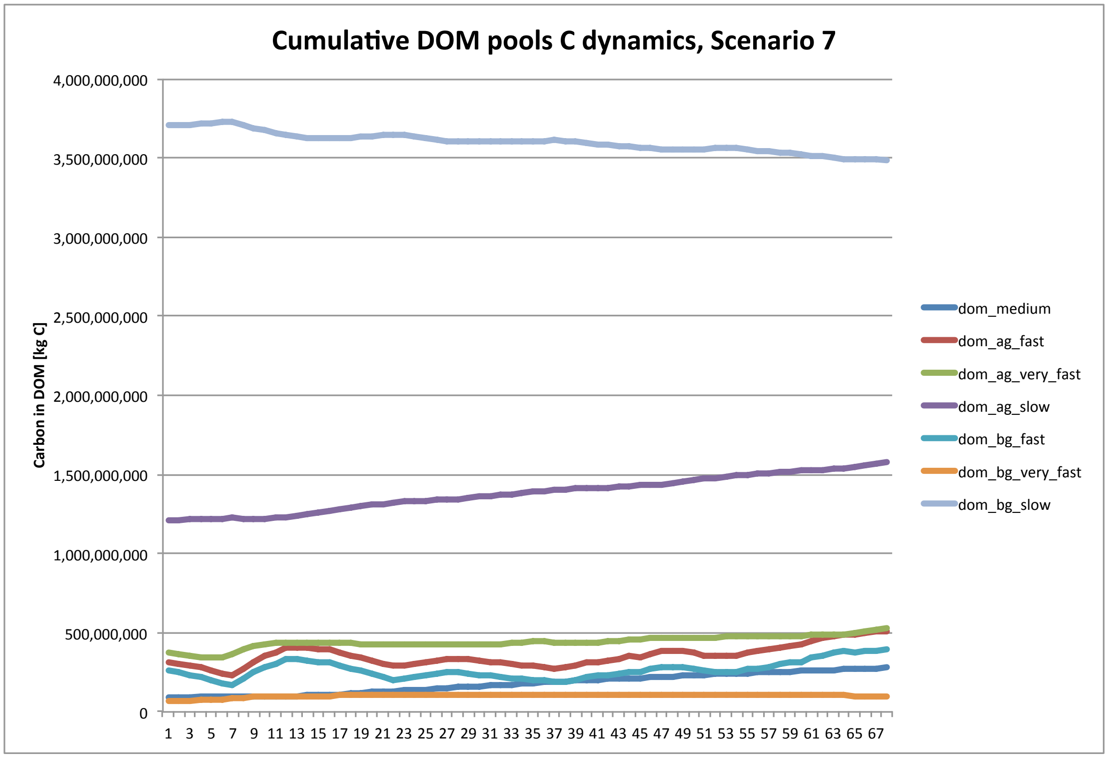

Figure E.11. Life cycle carbon dynamics in DOM pools, scenario 7

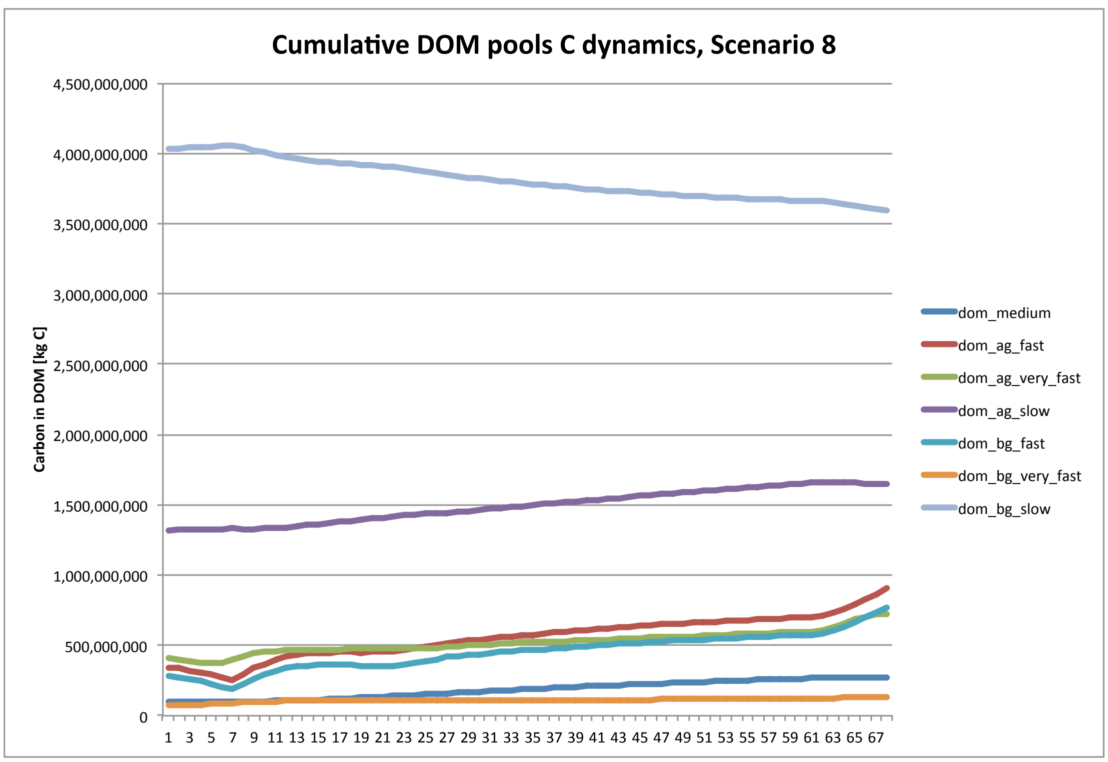

Figure E.12. Life cycle carbon dynamics in DOM pools, scenario 8

APPENDIX F. Examples of carbon dynamics for land parcels, irrigated and non-irrigated

Carbon dynamics in live biomass pools for a non-irrigated parcel in land unit LU1

Figure F.1 through Figure F.5 show examples of the carbon dynamics in live biomass pools for a non-irrigated land parcel in land unit 1. The treatments possible for this land unit type were S (single stem, non irrigated) and C (coppice, non-irrigated).

Figure F.1 represents scenario 1 where carbon is accumulating rapidly starting with year 1 until the first harvest year, which is eight years later as the treatment is S. At that point there is a sharp decline in the above-ground pools (stem, bark, branch and foliage) due to the removal of biomass through harvest, and in the below-ground pools (coarse and fine roots) due to their transfer to DOM pools as they are terminated in preparation for planting. All the following treatments chosen by the optimization model were also S, with harvests every 8 years.

Figure F.2 shows the carbon dynamics in live biomass pools for a non-irrigated land parcel in land unit 1, scenario 3. The optimization model chooses to apply first a treatment S, followed by two treatments C, two more treatments S, and the first two rotations of a treatment C. The increased carbon accumulation from one treatment to the next is attributed to the increase in biomass yield from one decade to the next.

The idle treatment can be also seen at the end of the project in scenario 8 (Figure F.3), when the model decided to not use this particular land parcel anymore.

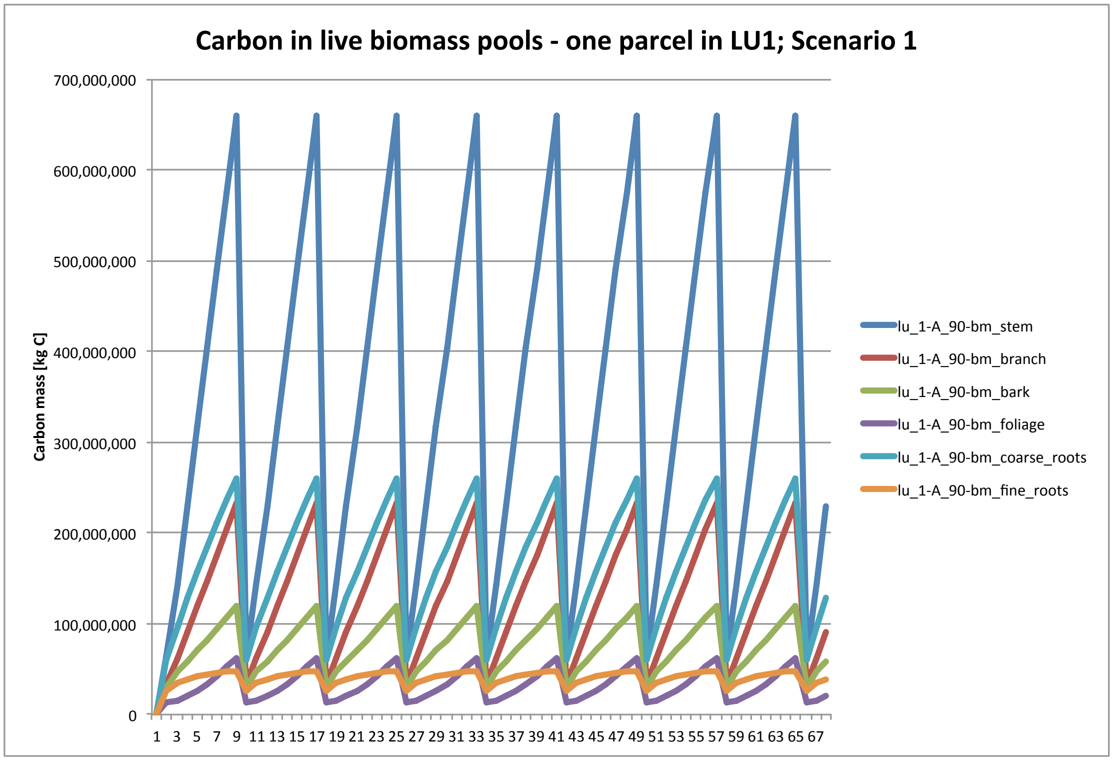

Figure F.1. Carbon in live biomass pools – one parcel in LU1; scenario 1

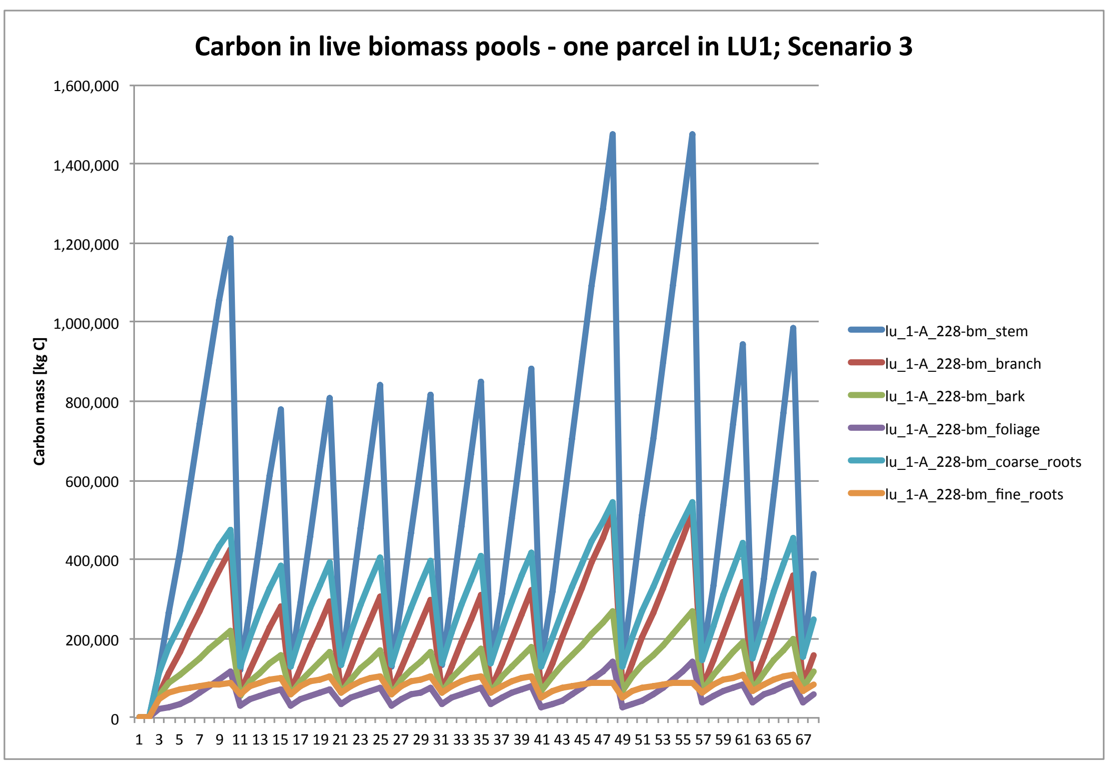

Figure F.2. Carbon in live biomass pools – one parcel in LU1; scenario 3

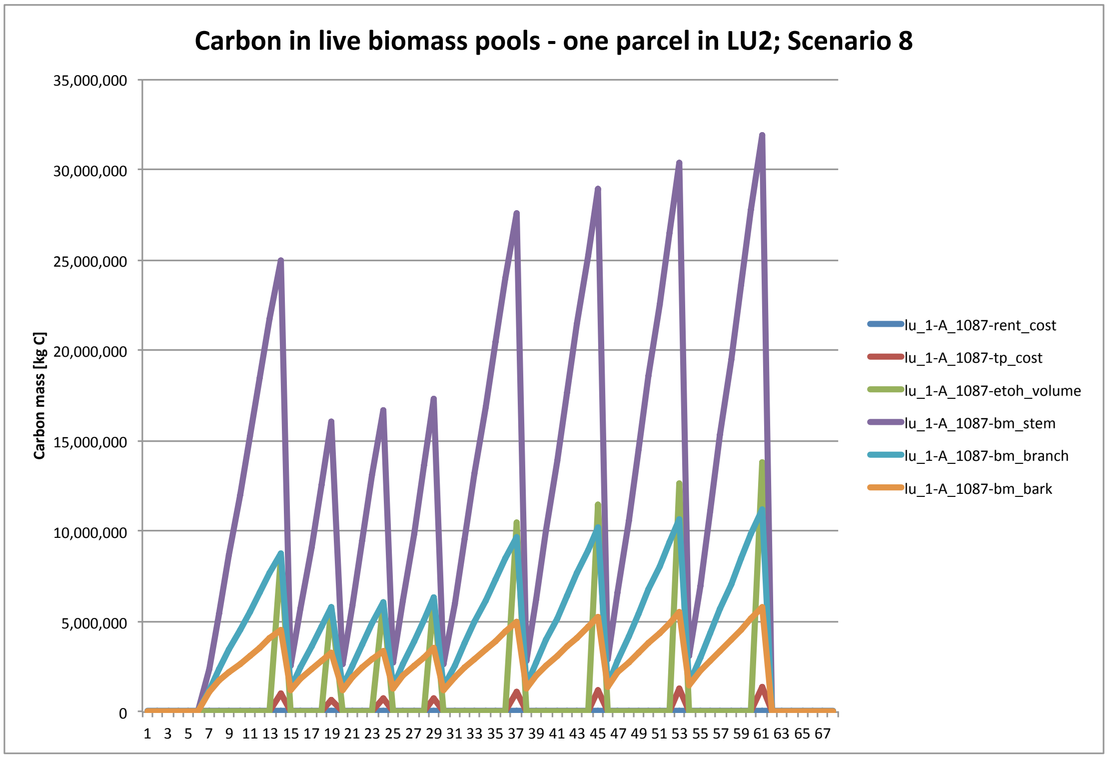

Figure F.3. Carbon in live biomass pools – one parcel in LU2; scenario 8

Carbon dynamics in live biomass pools for an irrigated parcel in land unit LU2

Figure F.4 and Figure F.5 depict examples of the carbon dynamics in live biomass pools for an irrigated land parcel in land unit 2. The treatments possible for this land unit type were Si (single stem, irrigated) and Ci (coppice, irrigated).

In scenario 3 (Figure F.4) the production plan starts with a treatment Si for 6 years, then all the following treatments were Ci. The continually increasing carbon accumulation from one treatment to the next is due to the decadal increases in biomass yield that are assumed in scenario 3.

In scenario 8 (Figure F.5) there are only two treatments, type Ci, after which the idle treatment was used so no more biomass was planted, and no more carbon accumulated in the live biomass pools.

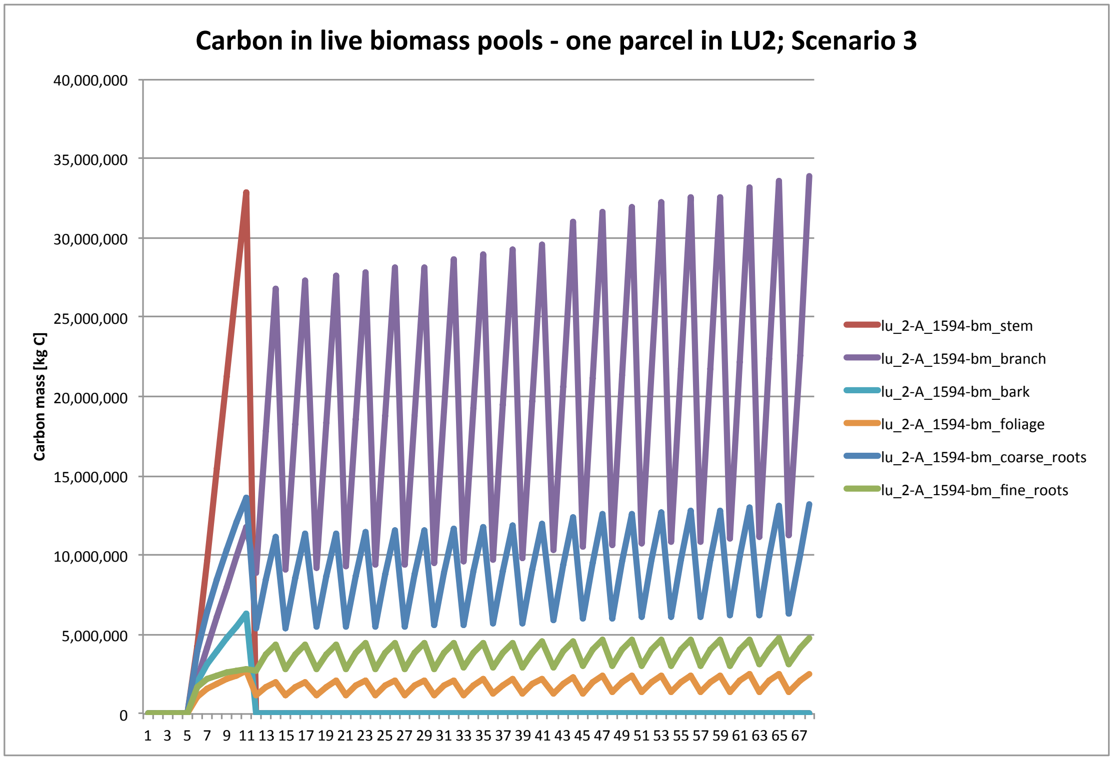

Figure F.4. Carbon in live biomass pools – one parcel in LU2; scenario 3

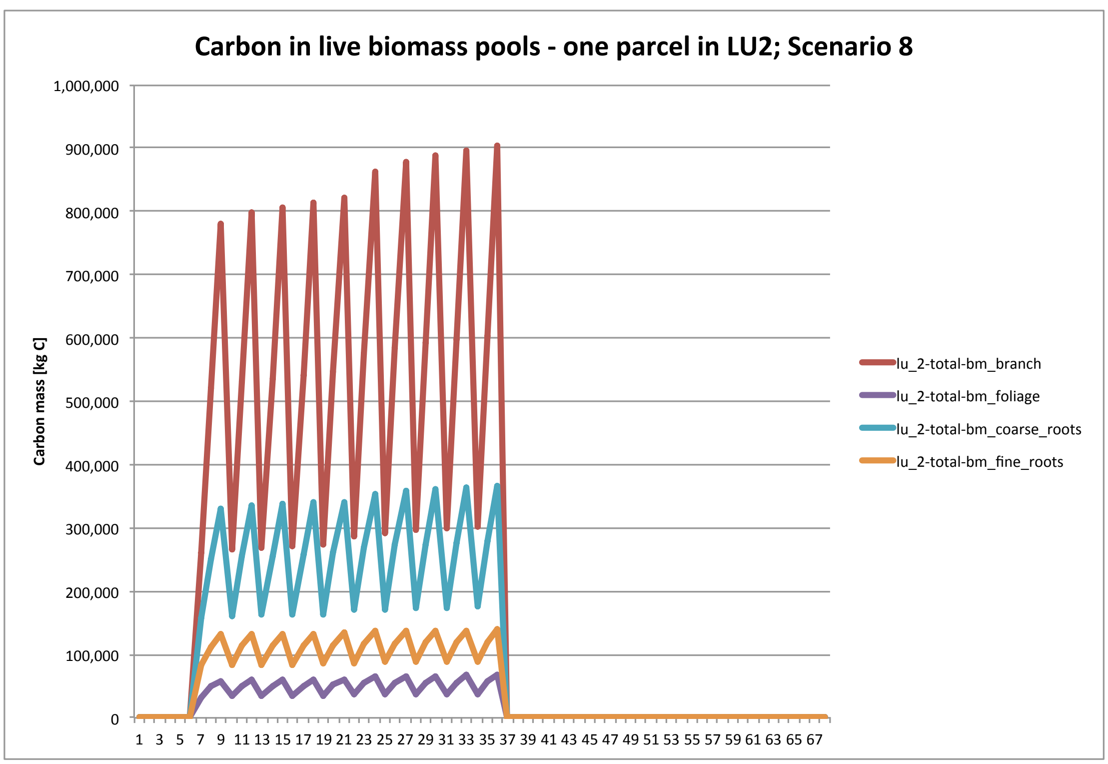

Figure F.5. Carbon in live biomass pools – one parcel in LU2; scenario 8

Carbon dynamics in DOM pools for a non-irrigated parcel in land unit LU1

The graphs in Figure F.6 and Figure F.7 show the carbon dynamics for a non-irrigated parcel in land unit 1. As expected, the carbon is continuously accumulating in the slow-decaying pools, both above- and below-ground, as well as in the medium-decaying pools. The spikes in carbon accumulation in DOM pools represent additions due to activities in harvest years, when some of the harvested live biomass is transferred to DOM pools, *i.e.* biomass that is not removed from site to be transported to the biofuel facility, and live roots which are terminated and assumed to become dead organic matter.

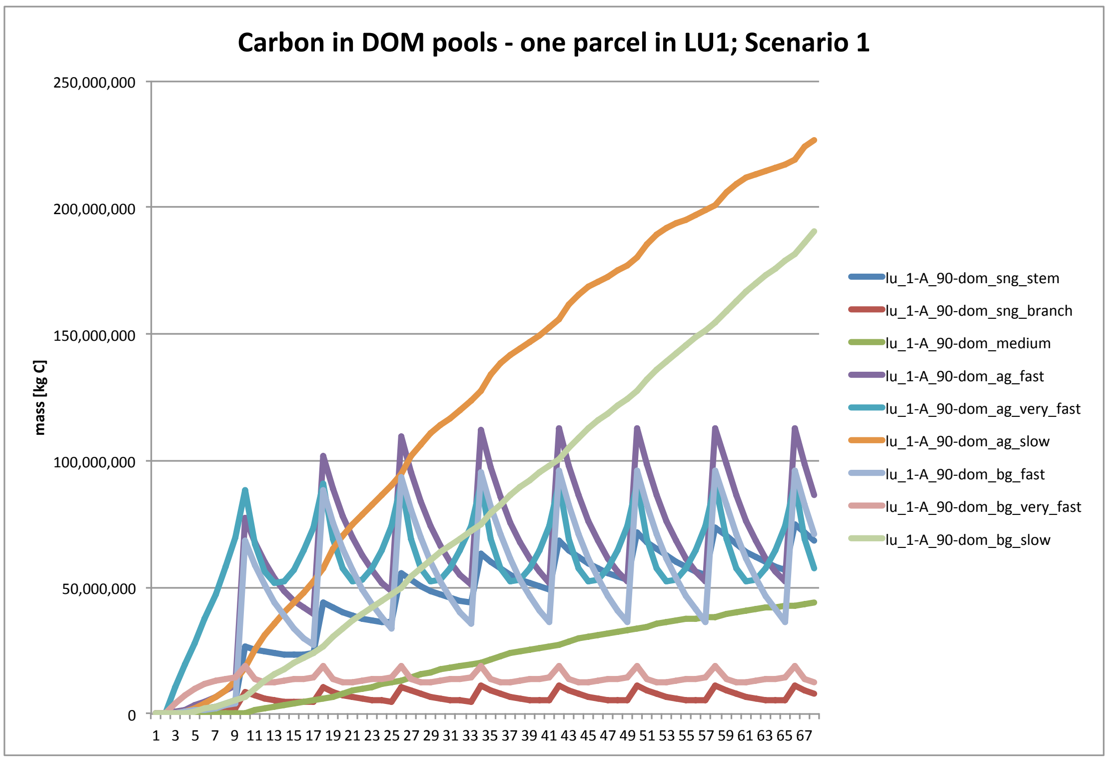

Figure F.6. Carbon in DOM pools – one parcel in LU1; scenario 1

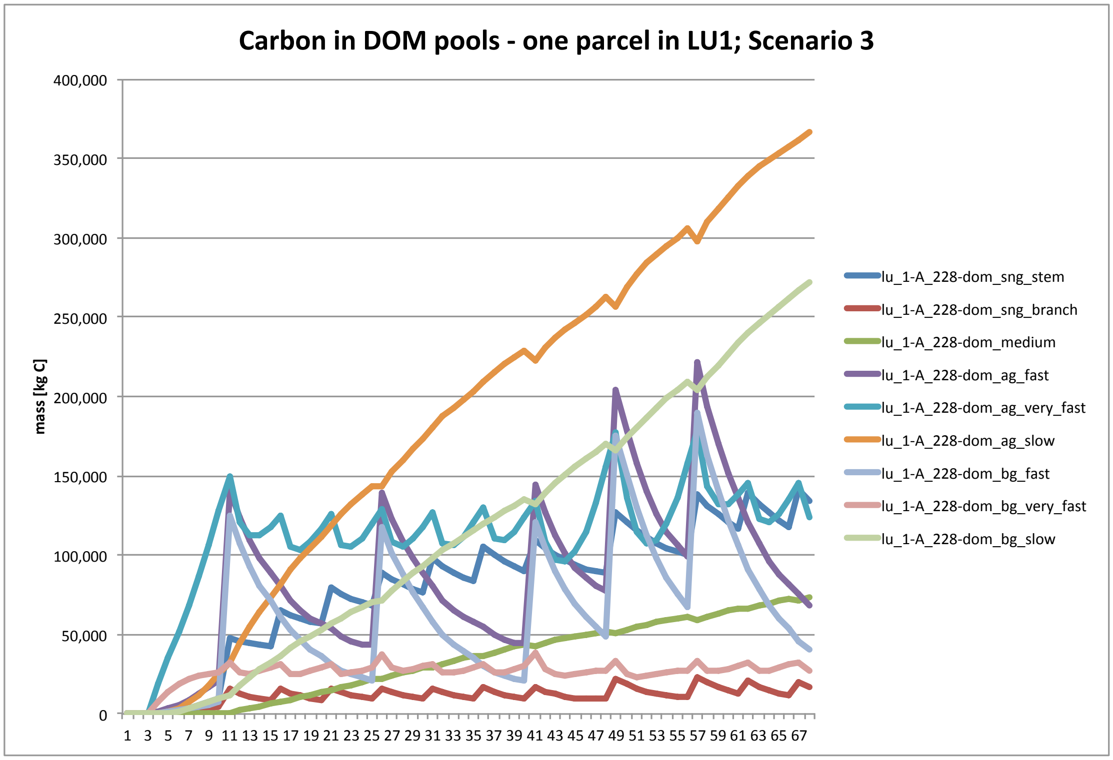

Figure F.7. Carbon in DOM pools – one parcel in LU1; scenario 3

Carbon dynamics in DOM pools for an irrigated parcel in land unit LU2

The carbon dynamic patterns for the irrigated parcel in land unit 2, scenario 3 (Figure F.8) are similar to those for a non-irrigated parcel.

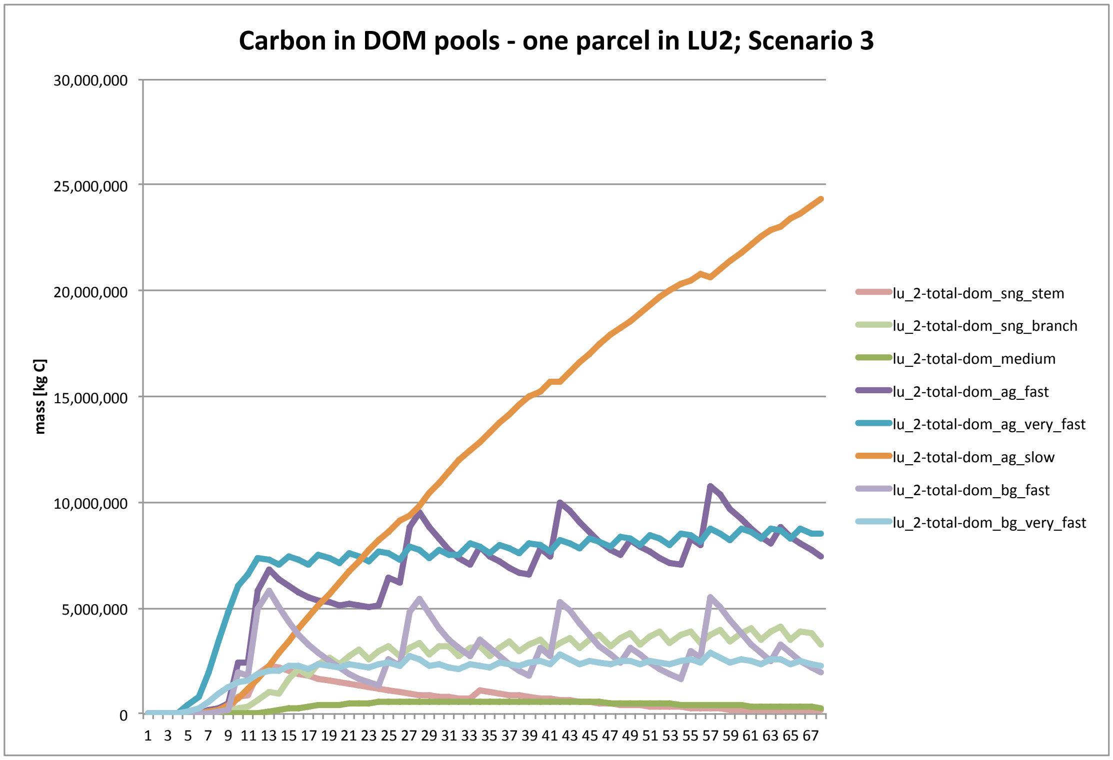

Figure F.8. Carbon in DOM pools – one parcel in LU2; scenario 3

In scenarios 1-4 the DOM graphs start from zero carbon, due to the assumption of no initial carbon stocks. In contrast, the carbon in DOM for scenarios 5-8 start out from a non-zero existing quantity (as shown in Figure F.9 and Figure F.10). The sequestration of carbon in DOM pools is the result of the balance between carbon the input from live biomass pools transfers, and the carbon output due to decay. The magnitude of the inputs versus the outputs determines whether the carbon dynamic in a particular pool is an increasing or a decreasing trend. In harvest years, the fast and very fast decaying pools experience sudden inputs (*i.e.* accumulations) of carbon due to transfer from live biomass pools, while the slow decaying pools show outputs (*i.e.* losses) of carbon due to the assumed impact of harvest activities. The idle treatment applied to the parcel shown in Figure F.10 starts with year 37 or so.

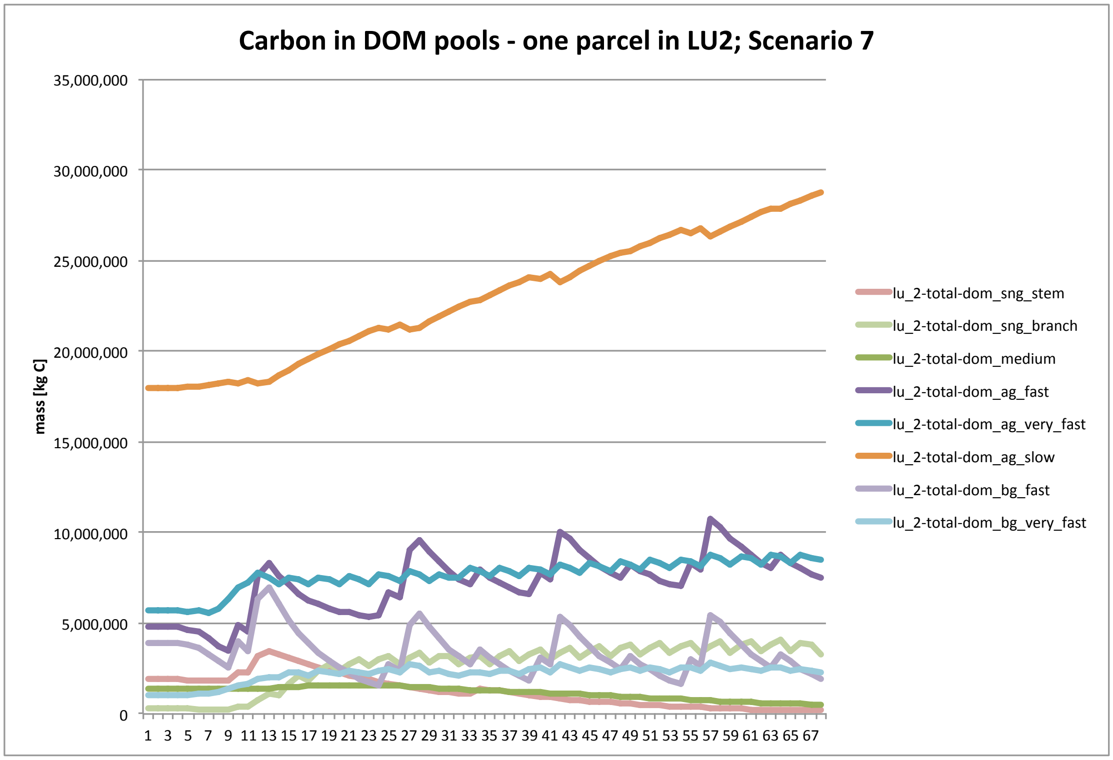

Figure F.9. Carbon in live biomass pools – one parcel in LU2; scenario 7

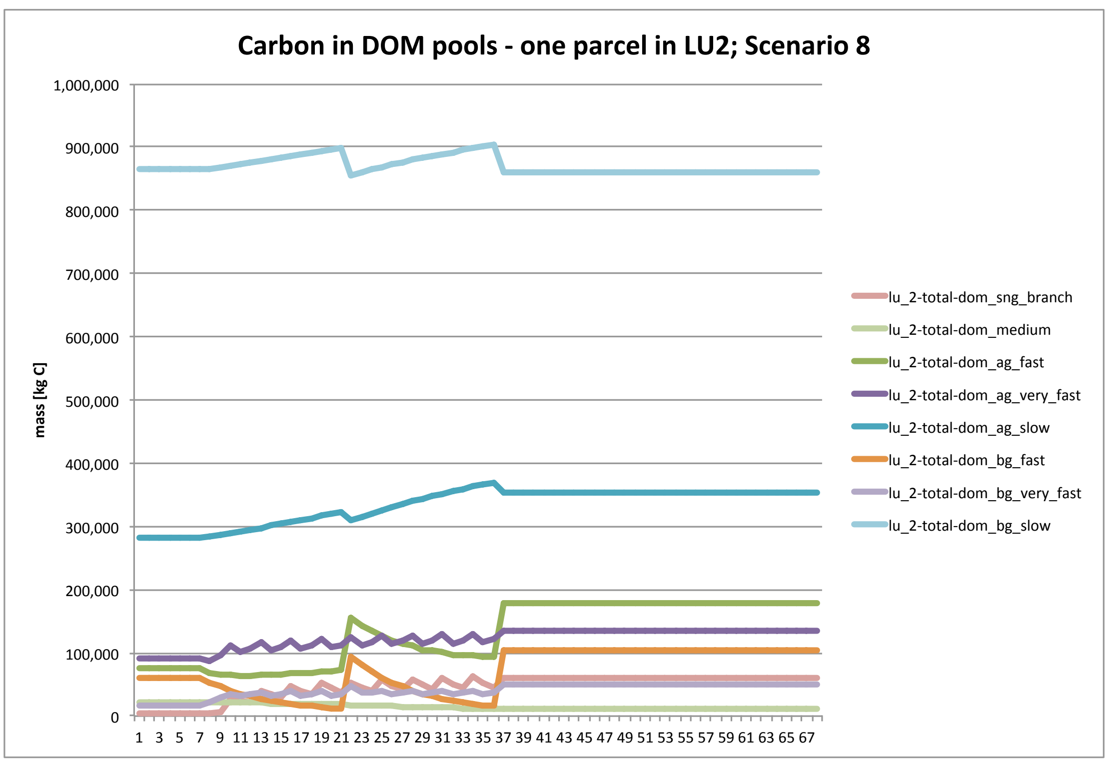

Figure F.10. Carbon in live biomass pools – one parcel in LU2; scenario 8

APPENDIX G. Two sample scenarios to illustrate positive and negative GHG savings

The four scenarios selected for the test case in CHAPTER 4 represent somewhat extreme conditions, showing either a very desirable GHG balance (*i.e.* biofuel project results in less emissions compared with the displaced fossil fuel system, see scenarios 1 and 2) or a very non-desirable GHG balance (the presence of the biofuel system is associated with more emissions than its absence, see scenarios 3 and 4).

To illustrate the other types of GHG balances that biofuel projects may result in, we selected two other scenarios:

- Scenario 5, which starts out with a large "carbon debt" (*i.e.* high loss of carbon from high initial carbon stocks at land-use change; higher GHG project emissions than non-project) but then it "recovers" and achieves a net negative emissions balance by the end of the project (less GHG project emissions than the non-project baseline)
- Scenario 6, which starts out in the first few years with a large sequestration of carbon in growing biomass but then the project produces more GHG emissions than GHG credits; by the end of the project there are more GHG emissions than the non-project baseline)

Table G.1. Values of input variables for test case scenarios

		Scenario 5	Scenario 6
Biomass MAI, treatment S	[t wood/ha/yr]	13.35	13.35
Biomass MAI, treatment Si	[t wood/ha/yr]	17.17	17.17
Biomass MAI, treatment C	[t wood/ha/yr]	13.35	13.35
Biomass MAI, treatment Ci	[t wood/ha/yr]	17.17	17.17
Biomass transportation distance	[km]	40	40
Conversion efficiency, white chips	[l EtOH/t wood]	299	299
Conversion efficiency, whole tree chips	[l EtOH/t wood]	268	268
Biorefinery capacity	[l EtOH]	227,124,707	227,124,707
Enzyme unit cost	[$/kg enzyme]	2.48	2.48
Electricity price	[$/kWh]	0.115	0.115

		Scenario 5	Scenario 6
Initial carbon stocks, vegetation above-ground	[t C/ha]	41.5	7.0
Initial carbon stocks, DOM	[t C/ha]	141.1	7.0
Loss of DOM carbon in harvest years: snag, medium, and slow-decaying pools	[%]	0%	0%
Loss of DOM carbon in harvest years: fast-decaying pools	[%]	0%	0%
Loss of DOM carbon in harvest years: very fast-decaying pools	[%]	0%	0%
DOM carbon loss at land-use change	[%]	25%	0%
EtOH production emissions	[kg CO_2/l EtOH]	0.4	0.8
Gasoline production emissions	[kg CO_2/ l gasoline]	3.3	2.9
Blend of ethanol in gasoline		E85	E85
Irrigation emissions	[kg CO_2/ha]	37	37

From the perspective of climate mitigation, scenario 5 (Figure G.1) is desirable only if the time horizon of the analysis is longer than approx. 60 years, when the project GHG balance is negative (*i.e.* less emissions than the displaced gasoline system by year 100). In contrast, scenario 6 (Figure G.2) is not a desirable mitigation option if the time horizon is longer than 85 years or so (*i.e.* more emissions than the displaced gasoline system by year 100).

These two scenarios illustrate how important is the selection of time horizon for which the project life cycle emissions balance is calculated.

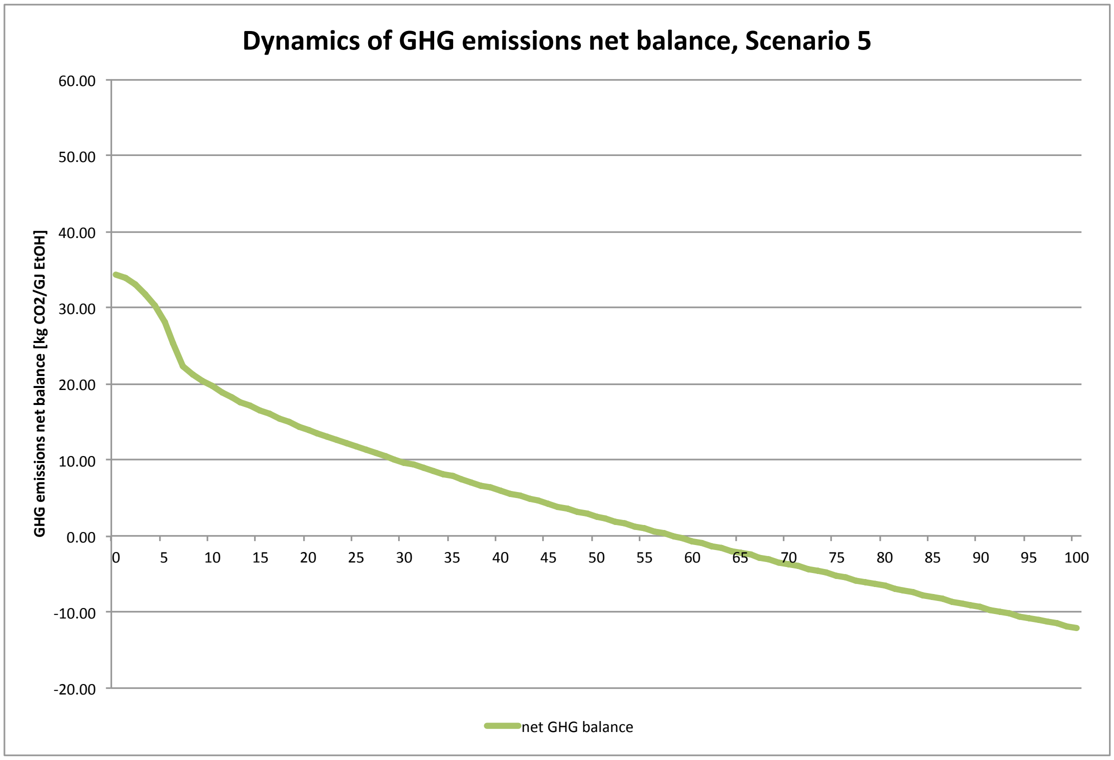

Figure G.1. Dynamics of GHG emissions balance dynamics, scenario 5, 100-year horizon

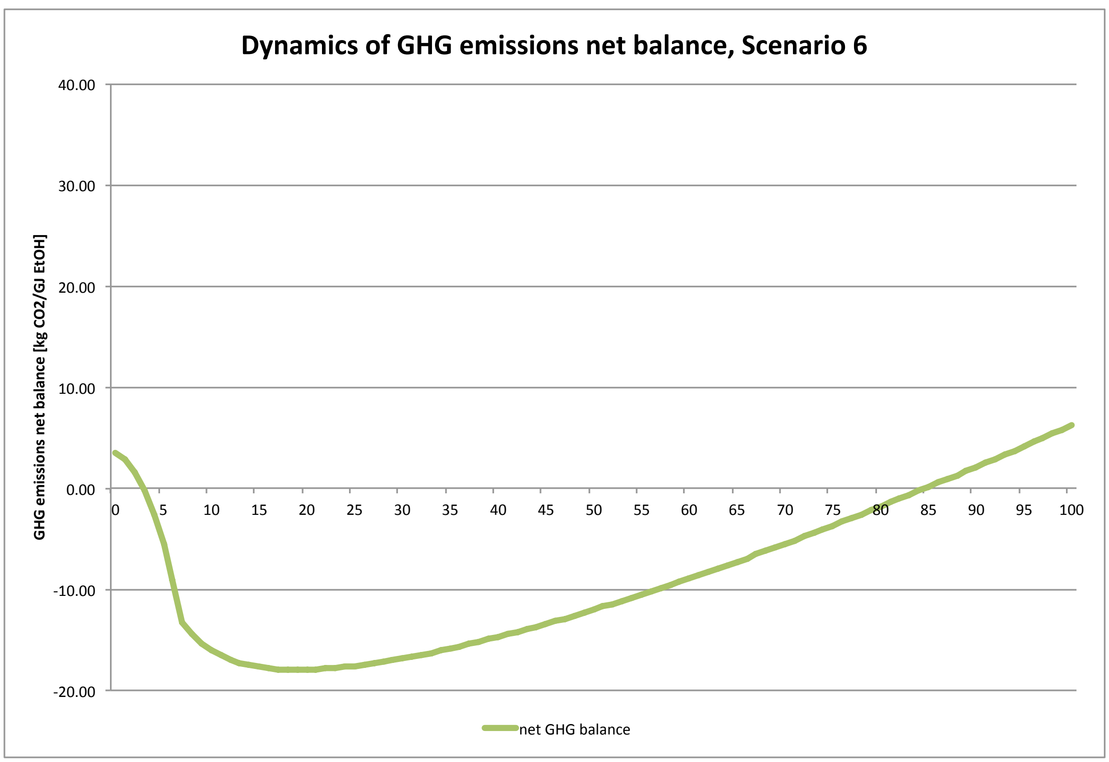

Figure G.2. Dynamics of GHG emissions balance dynamics, scenario 6, 100-year horizon

Table G.2. C3BO results: Plantation area needed and unit production costs for biofuel project, scenarios 5 and 6

	Scenario 5	Scenario 6
Plantation Area Needed [ha]	68,016	68,016
Biomass Harvest-Process Cost Total [$/t]	71.57	74.62
Land Rent Cost Total [$/t]	16.36	16.36
Biomass Transportation Cost Total [$/t]	29.37	29.37
Total Biomass Delivered Cost all lu's [$/t]	117.30	120.35
Total Biomass Delivered Cost [$/litre]	0.39	0.40
Refinery Cost [$/litre]	0.31	0.31
Total EtOH Production Cost [$/litre]	0.71	0.72

The GHG savings of the biofuel project (compared with the displaced fossil fuel system) are summarized in Table G.3. The biofuel project as considered in scenario 5 would result in 13% GHG savings compared with the displaced fossil fuel system (*i.e.* the baseline, the status quo). In other words, the existence of a biofuel project as described in scenario 5 would result in 13% less GHG emissions than what would have happened in its absence.

On the other hand, biofuel projects that are set up like scenario 6 would produce more GHG emissions (*i.e.* 7% more) than what would have happened in their absence (*i.e.* than the baseline).

Table G.3. Net GHG emissions and GHG savings of biofuel project

[kg CO_2/GJ EtOH] or [g CO_2/MJ EtOH]	Scenario 5	Scenario 6
Gross GHG emissions (GHG_PROJem)	83	90
Credit substitution fossil fuel (FOSScr)	-95	-84
Net GHG emissions (netGHG_PROJem Year 100)	-12	6
GHG savings of biofuel project	13%	-7%

SCENARIO 5

Net GHG emissions (netGHG_PROJem Year 100)	-12
Gross GHG emissions (GHG_PROJem)	83
Production emissions (GHG_PRODem)	98
Init. biomass removal emiss. (GHG_iLUCem)	19
Init. mech. activities emiss. (GHG_iMECem)	0
Proj. mech. activities emiss. (GHG_pMECem)	7
Init. transport emiss. (GHG_iTRNem)	0
Proj. transport emiss. (GHG_pTRNem)	15
Biorefinery infrastr. constr. emiss. (GHG_iINFRem)	0
Init. biorefinery activities emiss. (GHG_iREFem)	1
Init. conversion emiss. (GHG_iCNVem)	2
Init. lignin use credit (iLIGcr)	-1
Proj. biorefinery activities emiss. (GHG_pREFem)	56
Proj. conversion emiss. (GHG_pCNVem)	83
Proj. lignin use credit (pLIGcr)	-28
Carbon sequestration credit (CSEQcr)	-15
C from atmosphere credit (CATMcr)	-550
Proj. biomass removal emiss. (GHG_pBMHem)	278
DOM decay emiss. (GHG_DCAYem)	256
Credit substitution fossil fuel (FOSScr)	-95
Init. biofuel produced credit (iFOScr)	-4
Proj. biofuel produced credit (pFOScr)	-91

Figure G.3. GHG balance of the biofuel project, 100-year horizon, scenario 5

SCENARIO 6

Net GHG emissions (netGHG_PROJem Year 100)	6
Gross GHG emissions (GHG_PROJem)	90
Production emissions (GHG_PRODem)	147
Init. biomass removal emiss. (GHG_iLUCem)	3
Init. mech. activities emiss. (GHG_iMECem)	0
Proj. mech. activities emiss. (GHG_pMECem)	7
Init. transport emiss. (GHG_iTRNem)	0
Proj. transport emiss. (GHG_pTRNem)	16
Biorefinery infrastr. constr. emiss. (GHG_iINFRem)	0
Init. biorefinery activities emiss. (GHG_iREFem)	0
Init. conversion emiss. (GHG_iCNVem)	1
Init. lignin use credit (iLIGcr)	0
Proj. biorefinery activities emiss. (GHG_pREFem)	120
Proj. conversion emiss. (GHG_pCNVem)	179
Proj. lignin use credit (pLIGcr)	-59
Carbon sequestration credit (CSEQcr)	-56
C from atmosphere credit (CATMcr)	-572
Proj. biomass removal emiss. (GHG_pBMHem)	290
DOM decay emiss. (GHG_DCAYem)	226
Credit substitution fossil fuel (FOSScr)	-84
Init. biofuel produced credit (iFOScr)	-1
Proj. biofuel produced credit (pFOScr)	-83

Figure G.4. GHG balance of the biofuel project, 100-year horizon, scenario 6

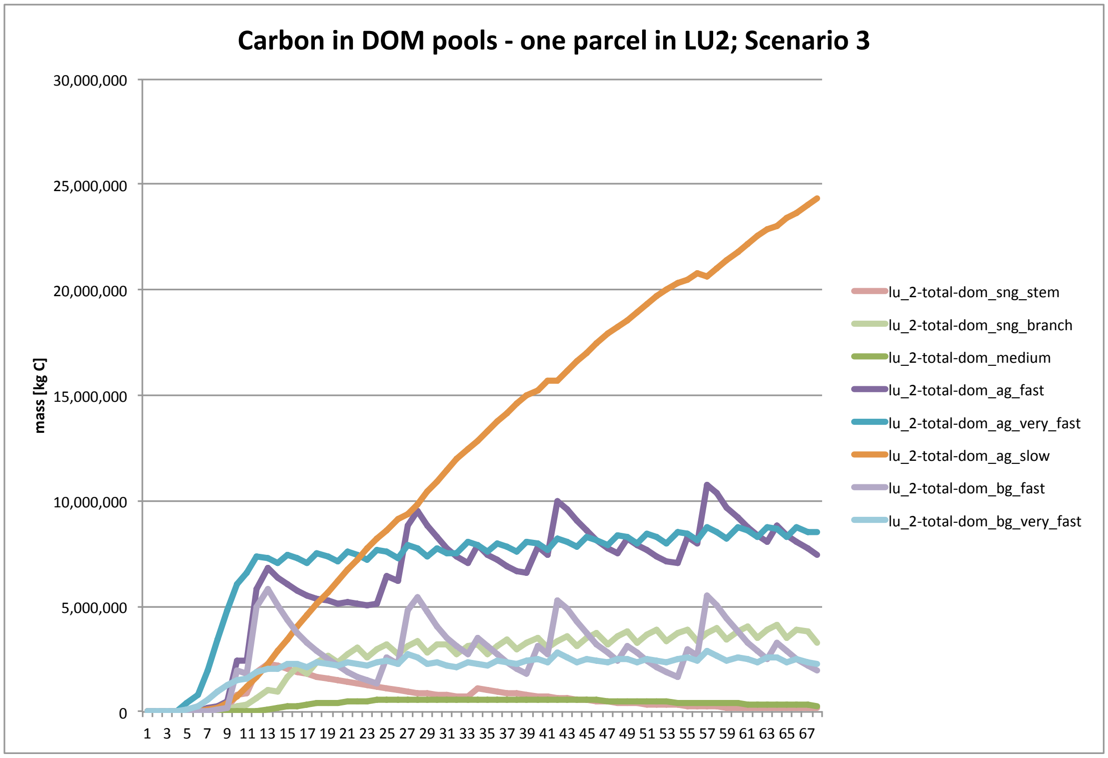

Figure G.5. Carbon in DOM pools – one parcel in LU2; scenario 3

In scenarios 1-4 the DOM graphs start from zero carbon, due to the assumption of no initial carbon stocks. In contrast, the carbon in DOM for scenarios 5-8 start out from a non-zero existing quantity (as shown in Figure G.6 and Figure G.7). The sequestration of carbon in DOM pools is the result of the balance between carbon the input from live biomass pools transfers, and the carbon output due to decay. The magnitude of the inputs versus the outputs determines whether the carbon dynamic in a particular pool is an increasing or a decreasing trend. In harvest years, the fast and very fast decaying pools experience sudden inputs (*i.e.* accumulations) of carbon due to transfer from live biomass pools, while the slow decaying pools show outputs (*i.e.* losses) of carbon due to the assumed impact of harvest activities. The idle treatment applied to the parcel shown in Figure G.7 starts with year 37 or so.

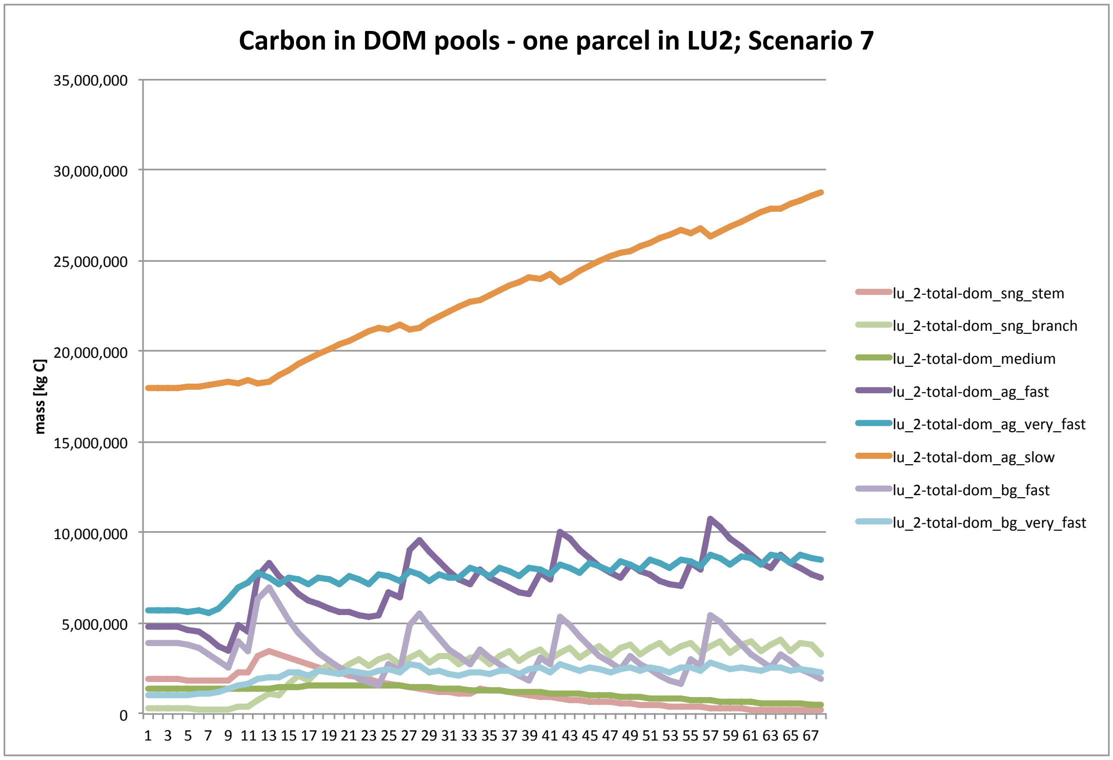

Figure G.6. Carbon in live biomass pools – one parcel in LU2; scenario 7

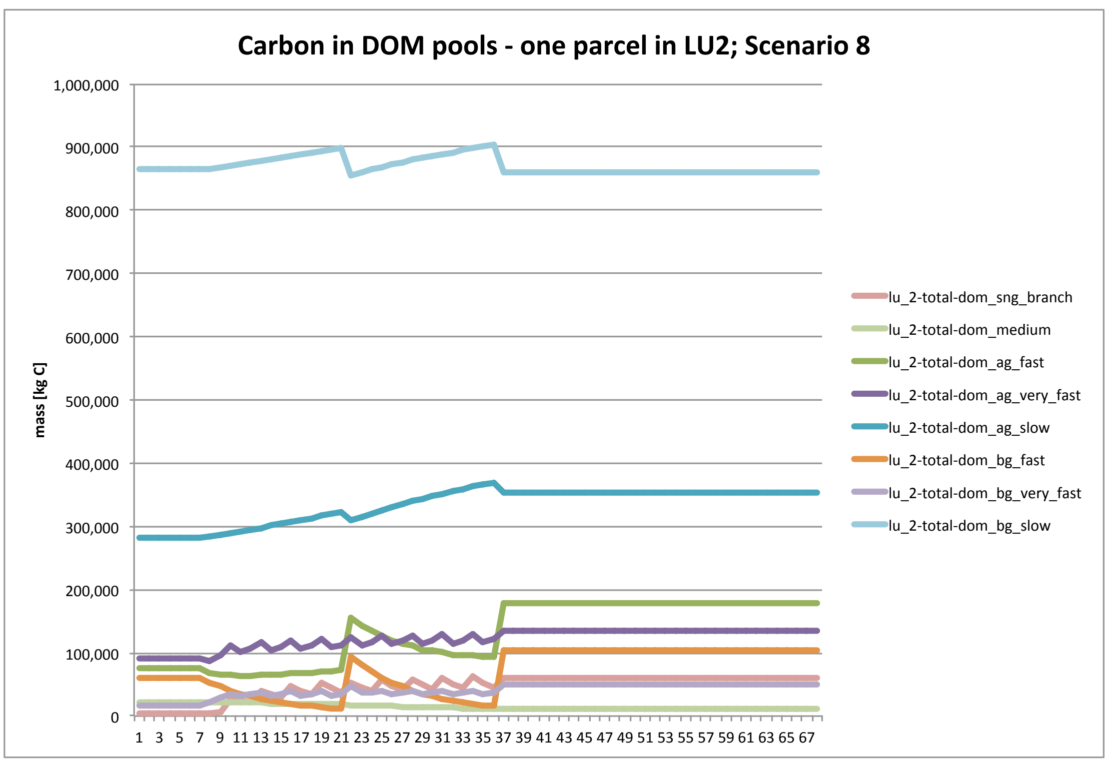

Figure G.7. Carbon in live biomass pools – one parcel in LU2; scenario 8

APPENDIX H. ANOVA tables and multiple linear regression models

Table H.1. ANOVA table for dependent variable Plantation Area

ANOVA Area	Df	Sum Sq	Mean Sq	F value	Pr(>F)	
Biomass yield; v01	1	3.10E+09	3.10E+09	230	<2.00E-16	***
Conversion efficiency; v03	1	1.46E+09	1.46E+09	108.2	4.56E-13	***
Biorefinery capacity; v04	1	4.15E+10	4.15E+10	3074.6	<2.00E-16	***
Residuals	41	5.53E+08	1.35E+07			

Table H.2. Linear regression model for dependent variable Plantation Area

	Estimate	Std. Error	t value	Pr(>|t|)	
(Intercept)	130703.085	10115.977	12.92	4.81E-16	***
Biomass yield; v01	-5103.47	382.077	-13.357	<2.00E-16	***
Conversion efficiency; v03	-235.584	28.153	-8.368	2.08E-10	***
Biorefinery capacity; v04	343.502	6.195	55.449	<2.00E-16	***
Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1 Residual standard error: 3674 on 41 degrees of freedom Multiple R-squared: 0.9881, Adjusted R-squared: 0.9873 F-statistic: 1138 on 3 and 41 DF, p-value: < 2.2e-16					

Table H.3. ANOVA table for dependent variable net GHG balance at 38 years

ANOVA NetGHG38	Df	Sum Sq	Mean Sq	F value	Pr(>F)	
Biomass yield v01	1	5331	5331	43.023	1.25E-07	***
Initial carbon stocks v06	1	212576	212576	1715.704	<2e-16	***
Loss of soil C (slow decaying pools) from mechanized activities in harvest years v07	1	5232	5232	42.227	1.51E-07	***
Transportation distance v02	1	291	291	2.348	0.13415	
Conversion efficiency v03	1	117	117	0.945	0.337381	
Ethanol production emissions	1	44061	44061	355.618	<2e-16	***

ANOVA NetGHG38	Df	Sum Sq	Mean Sq	F value	Pr(>F)	
v08						
Gasoline production emissions v09	1	1413	1413	11.402	0.001772	**
Carbon soil loss at land-use change v10	1	1886	1886	15.22	0.000402	***
Residuals	36	4460	124			

Table H.4. Linear regression model for dependent variable net GHG balance at 38 years

	Estimate	Std. Error	t value	Pr(>|t|)	
(Intercept)	-20.26425	37.24131	-0.544	0.589704	
Biomass yield v01	-6.31138	1.20883	-5.221	7.63E-06	***
Initial carbon stocks v06	4.08127	0.1066	38.285	<2e-16	***
Loss of soil C (slow decaying pools) from mechanized activities in harvest years v07	515.9861	64.7172	7.973	1.82E-09	***
Transportation distance v02	0.15063	0.04124	3.652	0.00082	***
Conversion efficiency v03	-0.11832	0.08662	-1.366	0.180446	
Ethanol production emissions v08	138.20829	7.24008	19.089	<2e-16	***
Gasoline production emissions v09	-103.7107	29.32783	-3.536	0.001138	**
Carbon soil loss at land-use change v10	89.94732	23.05561	3.901	0.000402	***
Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1 Residual standard error: 11.13 on 36 degrees of freedom Multiple R-squared: 0.9838, Adjusted R-squared: 0.9802 F-statistic: 273.3 on 8 and 36 DF, p-value: < 2.2e-16					

Table H.5. ANOVA table for dependent variable net GHG balance at 68 years

ANOVA NetGHG68	Df	Sum Sq	Mean Sq	F value	Pr(>F)	
Biomass yield v01	1	1517	1517	25.186	1.42E-05	***
Initial carbon stocks v06	1	91739	91739	1523.061	<2e-16	***
Loss of soil C (slow decaying pools) from mechanized activities	1	3306	3306	54.886	9.63E-09	***

ANOVA NetGHG68	Df	Sum Sq	Mean Sq	F value	Pr(>F)	
in harvest years v07						
Transportation distance v02	1	325	325	5.401	0.0259	*
Conversion efficiency v03	1	133	133	2.202	0.1465	
Ethanol production emissions v08	1	50214	50214	833.665	<2e-16	***
Gasoline production emissions v09	1	1539	1539	25.543	1.27E-05	***
Carbon soil loss at land-use change v10	1	350	350	5.813	0.0211	*
Residuals	36	2168	60			

Table H.6. Linear regression model for dependent variable net GHG balance at 68 years

	Estimate	Std. Error	t value	Pr(>|t|)	
(Intercept)	5.14784	25.96614	0.198	0.844	
Biomass yield v01	-4.15964	0.84285	-4.935	1.83E-05	***
Initial carbon stocks v06	2.57563	0.07433	34.652	<2e-16	***
Loss of soil C (slow decaying pools) from mechanized activities in harvest years v07	445.66612	45.12344	9.877	8.65E-12	***
Transportation distance v02	0.1426	0.02876	4.959	1.70E-05	***
Conversion efficiency v03	-0.11902	0.0604	-1.971	0.0565	.
Ethanol production emissions v08	145.31212	5.04808	28.786	<2e-16	***
Gasoline production emissions v09	-105.31001	20.44855	-5.15	9.49E-06	***
Carbon soil loss at land-use change v10	38.75699	16.0753	2.411	0.0211	*
Signif. codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1 Residual standard error: 7.761 on 36 degrees of freedom Multiple R-squared: 0.9857, Adjusted R-squared: 0.9825 F-statistic: 309.5 on 8 and 36 DF, p-value: < 2.2e-16					

Table H.7. ANOVA table for dependent variable net GHG balance at 100 years

ANOVA NetGHG100	Df	Sum Sq	Mean Sq	F value	Pr(>F)	
Biomass yield v01	1	553	553	15.314	0.000388	***

ANOVA NetGHG100	Df	Sum Sq	Mean Sq	F value	Pr(>F)	
Initial carbon stocks v06	1	54497	54497	1508.789	<2e-16	***
Loss of soil C (slow decaying pools) from mechanized activities in harvest years v07	1	2106	2106	58.3	4.90E-09	***
Transportation distance v02	1	323	323	8.933	0.005022	**
Conversion efficiency v03	1	131	131	3.638	0.064494	.
Ethanol production emissions v08	1	52729	52729	1459.829	<2e-16	***
Gasoline production emissions v09	1	1590	1590	44.019	9.94E-08	***
Carbon soil loss at land-use change v10	1	100	100	2.779	0.10418	
Residuals	36	1300	36			

Table H.8. Linear regression model for dependent variable net GHG balance at 100 years

	Estimate	Std. Error	t value	Pr(>|t|)	
(Intercept)	16.00611	20.10773	0.796	0.431	
Biomass yield v01	-3.11047	0.65268	-4.766	3.07E-05	***
Initial carbon stocks v06	1.90984	0.05756	33.181	<2e-16	***
Loss of soil C (slow decaying pools) from mechanized activities in harvest years v07	385.52736	34.94282	11.033	4.21E-13	***
Transportation distance v02	0.13855	0.02227	6.222	3.51E-07	***
Conversion efficiency v03	-0.11702	0.04677	-2.502	0.017	*
Ethanol production emissions v08	148.16762	3.90914	37.903	<2e-16	***
Gasoline production emissions v09	-106.06907	15.835	-6.698	8.19E-08	***
Carbon soil loss at land-use change v10	20.75231	12.44843	1.667	0.104	
Signif. codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1 Residual standard error: 6.01 on 36 degrees of freedom Multiple R-squared: 0.9885, Adjusted R-squared: 0.986 F-statistic: 387.7 on 8 and 36 DF, p-value: < 2.2e-16					