Ryan M. Bright

Environmental Systems Analysis of Road Transportation Based on Boreal Forest Biofuel

Case Studies and Scenarios for Nordic Europe

Doctoral Thesis

Ryan M. Bright

NTNU
Norwegian University of
Science and Technology

NTNU
Norwegian University of
Science and Technology

NTNU

Ryan M. Bright

Environmental Systems Analysis of Road Transportation Based on Boreal Forest Biofuel
Case Studies and Scenarios for Nordic Europe

Thesis for the degree of philosophiae doctor

Trondheim, July 2011

Norwegian University of
Science and Technology
Faculty of Engineering Science and Technology
Department of Energy and Process Engineering

NTNU
Norwegian University of
Science and Technology

Ryan M. Bright

Environmental Systems Analysis of Road Transportation based on Boreal Forest Biofuel: Case Studies and Scenarios for Nordic Europe

"My father rode a camel. I drive a car. My son flies a jet airplane. His son will ride a camel."

-   Anonymous Saudi Sheik, 1982

Preface

Global climate change and the onset of peak oil are formidable real world threats facing mankind. Road transportation is a sector in which both these threats require urgent attention and action. Biofuels are perceived as part of the technological solution. Unlike conventional oil-based transportation fuels, however, biofuels and their attributes are not uniform. Their raw material requirements and production methods vary substantially by region. Production of many of today's biofuels are energy and water intensive, few of which yield reductions in greenhouse gas emissions. The oil- and sugar-rich crops that are needed in their production need premium land to grow, turning fuel shortages into food scarcity.

Some regions posess significant raw material resources of the type not requiring use of premium agricultural lands, like Nordic Europe, for example – a region endowed with boreal forest resources. However, the technologies capable of producing biofuels from forest resources are commercially immature. These so-called "next-generation" biofuels are already learning to walk, but it might take two to three decades before they are free to run. In the meantime, many questions about their climate impacts remain unanswered. Never has the time been more ripe for industrial ecologists to play a role in biofuels research.

This thesis embraces systems thinking – a major tenet of industrial ecology research – drawing on a broad array of methods and systems analytic tools for the environmental assessment of "next generation" biofuels with significant emphasis on climate change. Climate change by nature is a global problem requiring local solutions. Through scenario analyses and case studies focusing on Nordic Europe, this thesis provides new insights into how regional transportation based on forest biofuels would play a role in mitigating climate change and fossil fuel dependency by analyzing, interpreting, simulating, and communicating the problem from multiple angles combining environmental, economic, technological, and policy perspectives.

Acknowledgements

First and foremost I wish to thank my supervisor, Anders H. Strømman, for his dedicated and invaluable mentorship and inspiration, without which I could not have been as productive over the past three years. Anders played a pivotal role in encouraging me to pursue the PhD, and along the way has continually challenged and pushed me to aim high in my research pursuits. I am extremely grateful for our many fruitful discussions, for his courtesy and respect for my own ideas, and for the liberty he has bestowed upon me to pursue them. Further, there have been other important past and present allies affiliated with the Industrial Ecology Program I wish to thank that provided additional intellectual support, most notably Troy R. Hawkins and Glen P. Peters. I sincerely appreciate them for their guidance, encouragement, enthusiasm, and eagerness to lend a helping hand in time of need or question.

I also wish to thank my parents, who have inspired me to follow their footsteps in academia and who will always serve as paramount mentors and role models in my life. They have provided sustained nourishment and support for over 30 years and have pushed me in striving to be the very best scientist and global citizen I can be. In addition, I express fond appreciation for my sister, Lisa, for her continual interest in my work and for her ability to keep me grounded in times of stress.

Last, but not least, I thank my best friend and partner, Bente Hammervold, whose love, wisdom, patience, and inspiration I will forever cherish. I am deeply indebted to the omnipresent solace she radiates in my direction that has enabled me to retain focus and perseverance throughout this scholastic endeavor.

Ryan M. Bright

Trondheim, July, 2011

List of Papers

Paper I

Bright, R. M. and A. H. Strømman, 2009. Life cycle assessment of second generation bioethanols produced from Scandinavian boreal forest resources: a regional analysis for Middle Norway. *Journal of Industrial Ecology*, 13(4): 514-531.

Paper II

Bright, R. M., Strømman, A. H., and T. R. Hawkins, 2010. Environmental assessment of wood-based biofuel production and consumption scenarios in Norway. *Journal of Industrial Ecology*, 14(3): 422-439.

Paper III

Bright, R. M. and A. H. Strømman, 2010. Incentivizing wood-based Fischer-Tropsch diesel through financial policy instruments: an economic assessment for Norway. *Energy Policy*, 38: 6849-6859.

Paper IV

Bright, R. M. and A. H. Strømman, 2010. Fuel-mix, fuel-efficiency, and transport demand affect prospects for biofuels in Northern Europe. *Environmental Science & Technology*, 44(7): 2261-2269.

Paper V

Bright, R. M., Strømman, A. H., and G. P. Peters, 2011. Radiative forcing impacts of boreal forest biofuels: a scenario study for Norway in light of albedo. *Environmental Science & Technology*, 45(17): 7570-7580.

Author's Contribution

All papers are co-authored. The author of the thesis has performed the following work for the papers:

Paper I: Data collection, modeling, analysis and writing.

Paper II: Data collection, modeling, analysis and writing.

Paper III: Data collection, modeling, analysis and writing.

Paper IV: Data collection, modeling, analysis and writing.

Paper V: Data collection, modeling, analysis and writing.

Table of Contents

Chapter 1: Introduction

1.1 Research Motives

1.1.1 Global Problem: The climate, energy, and transport nexus

Global climate change represents one of the largest threats facing mankind. In the EU and elsewhere, urgent actions are required to mitigate anthropogenic greenhouse gas emissions (GHG) for meeting a $2°C$ climate target, particularly those originating from the use of fossil fuels (Meinshausen et al., 2009; Roeckner, Giorgetta, Crueger, Esch, & Pongratz, 2011). In the EU, no other sector has experienced a GHG emission growth rate as high as that of the transport sector in recent decades, particularly road transport (DG TREN, 2009). These emissions can be viewed as a product of three components: i) the amount of the road transport activity that generates the emission, ii) the energy intensity of that activity, and iii) the GHG intensity of the energy that is being used. The amount of activity, or the overall demand for road transport, has greatly increased due in large part to tightly coupled growth in GDP and freight transport (DG TREN, 2009; European Commission, 2009). Additionally, little progress has been achieved in the way of reducing energy and GHG intensity (DG TREN, 2009; European Commission, 2009). The latter is owed to the fact that the road transport sector remains deeply dependent on fossil fuel use (97%, EU) which has additional negative implications for the security of energy supply (European Commission, 2009). This is not surprising since road transport infrastructures are essentially "locked-in" around the use of high density liquid energy carriers. Present and future alternatives over the near- and long-term time horizons appear to be limited to electricity, but it remains to be demonstrated as to whether heavy-duty freight or rural passenger road transport energy requirements can be met with battery-electric vehicles technologies (Ohlrogge et al., 2009; Savage, 2011). Thus it would appear that liquid fuels will continue to fuel mobility to some extent over the near- and medium-term horizon, and biofuels could provide a partial solution in the way of achieving the dual policy objectives of climate change mitigation and reduced fossil fuel dependency (Cruetzig & Kammen, 2010; Ribeiro et al., 2007).

1.1.2 Beneficial Biofuels – The food, energy, and environmental "trilemma"

Unfortunately, the implementation of poorly designed biofuel support policies in the EU preceded robust science surrounding their true benefits and external costs. A growing body of scientific literature suggests that many biofuels do not offer the proclaimed climate benefits (Farrell et al., 2006) and/or that they can pose additional threats to human health (Jacobson, 2007), food security (Pimentel et al., 2008; Pimentel & Pimentel, 2008), and other aspects of environmental quality (Bringezu et al., 2009). For most biofuels, a significant source of the proclaimed social and ecological impact can be attributed to the use of land required in biomass feedstock cultivation, either via poor land use management practice and the associated biogeochemical and biodiversity impacts – or via the displacement of food crops with energy crops, both of which may be either directly or indirectly attributed to their production (Crutzen, Mosier, Smith, & Winiwarter, 2008; Searchinger et al., 2008). A so-called food, energy, and environmental "trilemma" takes center stage in the recent biofuel debate, yet society cannot afford to miss out on the energy and climate mitigation opportunities when biofuels are "done

right" (D. Tilman et al., 2009). In other words, biofuels which are i) sourced from feedstocks not in competition with land for food production, and which are ii) derived from land that is managed responsibly and ecologically – can offer the desired energy and climate benefits without contributing to food insecurity and additional environmental degradation. These types of biofuels, often referred to as "next generation" or "advanced" biofuels, are derived from feedstocks mostly comprising agricultural residues, municipal solid wastes, and lignocelluloses produced on non-agricultural lands, and many have been shown to possess the beneficial benefits without imposing additional environmental and/or societal costs (Hill, Nelson, Tilman, Polasky, & Tiffany, 2006; Hill et al., 2009; Kim, Kim, & Dale, 2009; David Tilman, Hill, & Lehman, 2006).

The technologies to produce "beneficial" biofuels, however, remain unproven and immature at the commercial scale which carries two interesting connotations: a window of opportunity exists for researchers to comprehensively evaluate the environmental and GHG performance of novel conversion technologies before their market deployment and diffusion; however, the urgent mitigation action that is needed implies that governments may need to step in immediately and facilitate the deployment of technology "winners". In doing so, it makes sense first and foremost to think about the raw material production potential including the land use requirements. Taking this consideration into account, determining a sustainable quantity of biofuel to support can only be determined in a regional context (Stoeglehner & Narodoslawsky, 2009).

1.1.3 The Boreal Forest of Nordic Europe

The vast boreal forest resources of Nordic Europe – in particular Norway, Sweden, and Finland – provide a unique opportunity for expanded use of the resource as biofuel in those regions, evidenced by increasing trends in reserves of commercial growing stocks (FAO, 2007; UN FAO, 2010). At present there is little competition for forest biomass between the energy sector and the traditional forest products sector, and it is unlikely that such competition will occur in the immediate future (FAO, 2007). This implies that there may be some room for increased utilization of the resource for producing "beneficial" biofuels without compromising current societal demands for wood fibre and contributing to "indirect land use" or other market-mediated displacement effects. As such, several bioenergy support policies have or will soon be enacted throughout the region, such as Norway and Sweden's joint Green Certificate Scheme (Norwegian Ministry of Petroleum and Energy, 2010) and Finland's National Climate Action Strategy (Finnish Parliament, 2001). Further, Norway, Sweden, and Finland each have well-developed wood products and processing industries, and opportunities to use by-products from these industries warrants further exploration.

1.1.4 Local Solution: Forest-based Liquid Biofuels?

The regional "reserve" or surplus forest resource base presents a viable opportunity for the development of commercial biofuel industries throughout the region. On the surface, biofuels produced from biomass derived from responsibly managed forest areas appear to circumvent the food, energy, and environmental "trilemma" associated with many of today's crop-based biofuels (D. Tilman, et al., 2009). Relative to dedicated energy crops, forest biomass production requires little to no fertilizer application, and well-managed forests offer a great variety of ecosystem services such as biodiversity promotion, nutrient retention, and flood protection (Kauppi & Saikku, 2009). Further, the risks of losing permanent carbon stocks from direct land use change are low in the production of forest biomass, and risks of indirect land use change can be avoided

as long as stocks in surplus of those required to meet traditional societal wood demands are utilized.

1.2 Main Research Questions

The extent to which the boreal forest resources of Nordic Europe can be used in the production of liquid biofuels as a strategy to mitigate fossil fuel dependency and climate change stemming from road transportation is the subject of the academic research presented in this thesis. While the resource base appears to present an attractive opportunity to pursue such a strategy, major questions have yet to be answered from a scientific point of view. Much research has been done on single climate aspects of forest biofuels in general, but a broad analysis and assessment in a regional context is still missing for northern Europe. This thesis aims to fill this gap of knowledge by approaching the topic from multiple perspectives and analytic angles via the application of a variety of tools and methods of environmental systems analysis, techno-economic analysis, and climate impact modeling. The thesis makes extensive use of explorative scenarios and case studies, bridging environmental systems analysis with futures studies so that decision makers and stakeholders can begin a more informed discussion about the possibility of reaching targets and the necessary trajectories and challenges for developing and implementing technologies and policy.

The primary research questions in the thesis are:

- **Q1.** From a climate perspective, how does liquid biofuel produced from boreal forest biomass measure up to fossil fuel in the provisioning of a road transport service, and would any climate gains come at the expense of other environmental or human health impacts?
- **Q2.** What are the global climate and resource use implications when production of forest biofuel is scaled up commercially in a regional context?
- **Q3.** What are the costs, both to the public and private sectors, of deploying novel forest biofuel production technologies in the near term?
- **Q4.** How would other factors beyond changing the carbon intensity of the fuel supply affect resource use and climate impacts of road transportation throughout the region?

Environmental systems analysis research of biofuel is a relatively young and rapidly advancing field, which means staying at the forefront of the research frontier is a challenge. In the course of creating this thesis, further questions arose regarding the climate impacts of forest biofuels:

- **Q5.** What are the climate implications attributed to the use of forest biofuel when forest carbon cycle dynamics and biogeophysical land use effects like surface albedo changes are considered?

The primary questions posed above are each addressed in separate chapters, which are then decomposed into more detailed questions and specific research objectives. The structure of the thesis is presented in Chapter 1.7.

1.3 Research paradigms

1.3.1 The Need for Interdisciplinary Research

As societal problems increase in complexity, so too do the efforts required to mitigate them. One cannot ignore the possibility that, all too often, solutions to many problems generate new problems, adding to complexity (Tainter, 1988, 2000). To reduce the risk of creating new problems, new research paradigms ought to be embraced. While disciplinary research and teaching dominate in academia, there is a growing call for more interdisciplinary research (Becker, 2002; Bruce, Lyall, Tait, & Williams, 2004; Morse, Nielsen-Pincus, Force, & Wulfhorst, 2007; National Academy of Sciences, National Academy of Engineering, & Institute of Medicine of the National Academies, 2005). This is understandable, since many achievements of disciplinary research have been accompanied by negative outcomes such as growing economic inequality, continuing food insecurity, and environmental degradation (Jabbar, Saleem, & Li-Pun, 2001). Sometimes, the problems leading to these outcomes are not easily expressed in terms of disciplinary knowledge, and addressing them requires creating inherently useful forms of knowledge, rather than creating new disciplines, which may require reaching across disciplines for a particular purpose rather than filling in the gaps between them (Becker, 2002; Robinson, 2008).

It is becoming increasingly apparent that improving human well-being and promoting sustainable development requires a holistic and integrated approach that transcends disciplines. Interdisciplinary research starts from the premise that any "problem" or complex reality can be viewed and interpreted from a variety of non-equivalent perspectives, and within each perspective, a problem or reality can be understood from a range of spatial and temporal scales (Jabbar, et al., 2001; Rosenfield, 1992; Wrisberg et al., 2002). The perfect example is climate change, which can be described and understood from different disciplinary perspectives at global, regional, and local levels over varying time periods. Within each perspective and scale, those contributing to – and those suffering the consequences of – may identify different elements, use different indicators, and draw different conclusions. The perspectives that are considered, how they are incorporated in research, and the scale of research all determine the outcomes. Interdisciplinary research may help in integrating various perspectives and scales, and it itself is shaped by the nature of the problems to which it is addressed.

The field of Industrial Ecology and sustainability research represents a paradigm case of interdisciplinary research because of its innately complex, multi-faceted, and problem-based focus (Allenby, 1999; Klein, 2004; Lowe & Phillipson, 2009; Robinson, 2004, 2008). Being problem-based raises pertinent questions about how problems are defined and studied, particularly when they have to do with managing the interface between academic norms and standards, and the urgencies and practical politics of the problems that are chosen. The challenge lay in the choice of research problems that will be both academically and socially fruitful. For example, too heavy emphasis on the former leads to research that, while successfully addressing problems that make an important literature contribution within a particular field of study, may be of limited value or interest outside of academia. On the other hand, too much emphasis on the latter can lead to work that is indistinguishable from consulting or pure advocacy work (Robinson, 2008).

The point is that being problem-driven means starting from a societal problem or concern, yet in order to find good balance between academically and socially beneficial research, the problem

must be translated into a form that is open to interdisciplinary research, which often gives rise to new methods and tools that combine theory and practice in innovative ways. These innovations in theory and method are less in the direction of linear, step-by-step approaches, and more in the direction of heuristic search strategies for possible problem solutions (Becker, 2002). In Industrial Ecology or sustainability-related research, problems or issues are addressed by examining them from multiple perspectives involving aspects of the environment, the economy, society, and technology. Problems are approached – not by reacting to a specific part – but by viewing them as parts of an overall system. While fundamental to the field of Industrial Ecology, in general this sort of "systems thinking" – focusing on cyclical rather than linear cause-effect relationships – is becoming increasingly embraced in problem-oriented, interdisciplinary research.

1.3.2 A Need for a Systems Approach and Life Cycle Thinking

To think in "systems" means to operate within a framework that is based on the belief that the component parts of a system can best be understood in the context of relationships with each other and with other systems, rather than in isolation. In other words, in contrast to reductionism, systems thinking means viewing systems in a holistic manner. Embracing systems thinking can help one map and explore dynamic complexity, acquiring a unique perspective on reality which may sharpen an awareness of whole and how the parts within the whole interrelate.

Biofuels are specific technologies proposed as one solution to the problem (climate change and fossil-dependent road transport), and examining them in the context of the larger socio-economic and environmental "system" to which they are embedded can help to ensure that any decision supporting their development does not generate new problems or shift them to other areas and/or points in time. One concept of systems thinking particularly well-suited to address aspects of problem-shifting in the assessment of biofuels is "life-cycle thinking" which reflects the consideration of "cradle-to-grave" implications of biofuel production and use (Bringezu, et al., 2009; Cruetzig & Kammen, 2010; Kammen, Farrell, Plevin, & Jones, 2008). Life cycle thinking can help illuminate the many wide and interrelated factors in need of consideration when deciding on the relative merits of pursuing one biofuel over another. A number of analytical tools embracing concepts of "systems" and "life cycle" thinking have been (and continue to be) developed and refined over the last several decades, and those developed for use specifically in the environmental management context comprise many of those employed in Environmental Systems Analysis research.

1.4 Environmental Systems Analysis

There is no single, strict definition of environmental systems analysis, but some have defined it as:

> *…a subject area which seeks solutions to environmental problems from the point of view of technical supply systems, with the research goal being to offer insight into those technical solutions which are more sustainable than today's and for ways of changing these technical systems so that they can better meet the limitations set by the availability of natural resources and environmental degradations* (Chalmers University of Technology, 2011).

Others have defined it more generically as:

…a quantitative and multidisciplinary research field aimed at analyzing, interpreting, simulating, and communicating complex environmental problems from different perspectives (Wageningen University, 2011).

By these very definitions research in Environmental Systems Analysis is problem-oriented and interdisciplinary as it combines ecological, economic, technological, and policy perspectives in order to develop new insights into the causes, effects, and potential solutions (like biofuels) to complex environmental problems (like climate change). Often, it makes use of a systems perspective, drawing on a set of methods and tools for the environmental assessment of anthropogenic systems. These tools are structured vehicles for reasoning, analysis, and communication, and are becoming increasingly available for use by companies, governments, authorities, and others – including academics (Finnveden & Moberg, 2005). Some have made the distinction between tools which focus on procedural aspects that guide the way to reach a decision, or "procedural tools", and those which provide technical information as to the consequences of a choice, or "analytical tools" (Udo de Haes, Heijungs, Huppes, van der Voet, & Hettelingh, 2000; Wrisberg, et al., 2002). In the thesis, Environmental Systems Analysis is viewed as a learning process – as opposed to data gathering or decision making – for the production and effective communication of arguments relevant in a particular context.

A brief introduction to the specific analytical tools of Environmental Systems Analysis in addition to other research tools applied in the thesis are presented in the following subsections of this chapter. Only a brief synopsis related to their general utility including major methodological strengths and weakneseses is provided here. They are described more comprehensively – together with motives for their application – in Chapters 2 - 6.

1.4.1 Life Cycle Assessment (LCA)

Life Cycle Assessment (LCA) can succinctly be described as a micro-level decision-analytic tool for system-level comparisons of products[1], applied with the goal of specifying the environmental consequences of those products from "cradle-to-grave" – or over their entire life cycle from resource extraction to disposal. It is often used for measuring and comparing the impacts of technologies and product systems in the promotion of sustainable development. LCA has been recognized since the 1990s as a valuable decision-support tool, exemplified by its international standardization (ISO 14040:2006; ISO 14044:2006) and application in third-party certified product labeling schemes (i.e., for ISO Type III labels like Environmental Product Declarations). According to the ISO 14040 and 14044 standards, the four distinct analytic phases of an LCA involve: i) defining the goal and scope of the study, setting its context and explaining how and to whom results are to be communicated; ii) compiling material and energy inputs and outputs in the product system; iii) evaluating impacts associated with these inputs and outputs; and iv) interpreting results, including the evaluation of the study considering completeness, sensitivity, and consistency – in addition to formalization of conclusions and outlining of recommendations and limitations.

The tool's strength lay in its holistic scope – the comprehensive evaluation of upstream and downstream flows within a product system related to that product's function – which avoids

---

[1] "Products" refer to goods, services, or other types of functions

problem shifting from one stage in the life cycle to another, from one environmental problem to another, and from one location to another.

1.4.1.1 LCA Weaknesses

While the comprehensiveness of LCA with respect to environmental impact connected to a function is seen as a major strength, it may also be viewed as a weakness: the all-inclusive chain of processes that must be considered and the associated acquisition of input and output data (LCI) is time consuming and expensive (Guinée et al., 2002; Udo de Haes, Heijungs, Suh, & Huppes, 2004). Other aspects of LCA which may be seen as limitations are that it is neither site-specific nor dynamic, meaning that it does not directly consider future changes in technology and demand, rebounds, and other induced effects (Wrisberg, et al., 2002). In other words, results have a low spatial and temporal resolution, and social and economic aspects are not taken into account (Owens, 1997b; Udo de Haes, et al., 2004).

1.4.2 Environmentally-extended Input-Output Analysis (EE-IOA)

Environmentally-extended input-output analysis (EE-IOA) has been recognized since the 1970s as a macro-level tool capable of attributing environmental impact and resource use of economic activity to final demand in a consistent framework (Wiedmann, 2009). EE-IO models can be built to increase one's understanding of the generation of negative externalities resulting from economic activity, and subsequently, how conventional input-output computations can answer unanswered questions about the undesirable environmental effects of modern technology and unconstrained economic growth. They rely on aggregate sector-level data to describe how much environmental impact can be attributed to a sector in an economy, and how much each sector purchases from other sectors when a complete basket of goods and services (i.e., multiple functions) must be provided (Weber, Hendrickson, & Matthews, 2010). Such analysis can account for long production chains (for example, building an automobile requires energy, but producing energy requires vehicles, and building those vehicles requires energy, etc.), and, from a life cycle perspective, this somewhat alleviates the scoping problem of process LCA by accounting for impacts of the full upstream supply chain (Murray, Wood, & Lenzen, 2010).

An appealing aspect of EE-IOA lay in the fact that the input-output table is one of the only publically available statistics based on a well-established method of compilation that reveals the structure of inter-industry interdependence at the national level (Suh & Kagawa, 2009). This contrasts with LCA and the difficulties of obtaining reliable data from industry and the costs of collecting such data, although this has been improved in recent years through the development of comprehensive LCI databases like Ecoinvent (Ecoinvent, 2009) – and will continue to be improved upon implementation of the International Reference Life Cycle Data System (ILCD) (Heinrich, 2010).

1.4.2.1 EE-IOA Weaknesses

While the system boundary completeness/scoping limitation issues of process-LCA can be overcome in EE-IOA, it comes at the expense of having to rely on aggregate sector-level average data (Murray, et al., 2010). This data are often not very representative of the specific subset of the sector relevant to a particular product and therefore may not be suitable for evaluating the environmental impacts of products. Other limitations arise from its use in dynamic applications, or when decision support at the macro-level requires an understanding of inducement and/or

rebound effects resulting from changes in technology, i.e., structural changes in other economic sectors leading to unwanted environmental effects (Vögele, Kuckshinrichs, & Markewitz, 2009). This is due to the methodological framework itself and its reliance on constant technical coefficients and linear production functions that are not possible to avoid, which means that when a new technology allows either input substitution or more efficient use of inputs, impacts to supplying industry sectors may be misrepresented (Ardent, Beccali, & Cellura, 2009).

However, the linearity of the mathematical relationships in IO-based models can also be seen as strength in applications that require easy comprehensibility of modeling interactions and high transparency of cause-effect relationships. For these reasons, it is perceived as an invaluable Environmental Systems Analysis tool of Industrial Ecology research.

1.4.3 Multi-region Input-Output Analysis (MRIOA)

In recent years, increased globalization of production networks has stimulated research into the effects of trade on the environment. Applications of multi-regional input-output (MRIO) modeling frameworks have been instrumental in such research (G. P. Peters & Hertwich, 2009). Capturing the environmental effects of trade necessitates a modeling framework with expansive spatial resolution to account for structural differences in region-specific production technologies. In an MRIO model, a sectoral representation of technology structure is distinguished between those of an exporting region and those of an importing region, and, when coupled with resource use or environmental externality data, may be employed to quantify environmental impacts embodied in trade.

1.4.3.1 MRIO Weaknesses

The MRIO framework requires considerable data, much of which is neither readily available (G. P. Peters & Hertwich, 2009) nor in the correct format (Wiedmann, 2009) – both of which can lead to additional sectoral aggregation error and uncertainty (Lenzen, Pade, & Munksgaard, 2004).

1.4.4 Integration of Environmental Systems Analysis Tools

Tools of environmental systems analysis can be combined to avoid problem shifting since no single tool is capable of addressing all relevant questions nor is it capable of describing all types of problem shifting (Finnveden & Moberg, 2005; Udo de Haes, et al., 2004; Wrisberg, et al., 2002). Another prominent reason for combining tools is to maximize strengths and minimize weaknesses. For example, EE-IO/MRIOA's reliance on sector-level averages can be overcome through disaggregation of the sector producing the particular product of interest into distinct processes to provide more resolution, or via augmentation with additional technological data derived from bottom-up engineering models and/or physical life cycle inventory data. This is often referred to as "hybrid" LCA (Joshi, 2000; Suh & Huppes, 2002; Treloar, 1997; H. C. Wilting, 1996). Thus, the adjoining of product-specific life cycle inventory data of the foreground system with input-output data representing the background system both reduces aggregation error and expands the system boundaries. Further, the data-intensive weaknesses of LCA, notably the compilation of life cycle inventory (LCI) data comprising the background system, may be overcome by supplementing with public economic input-output data which are readily available from national statistical agencies.

Regardless of whether IO-based or LCA tools are used in conjunction or isolation, they can best be characterized as belonging to one encompassing family of "fixed coefficient" tools for

environmental analysis (Udo de Haes, et al., 2000), and as such, are very well-suited to grasp the environmental consequences of changes in the current production and consumption structure (Wrisberg, et al., 2002).

1.4.5 Economic Tools

Technological decision making often requires the application of complimentary analytical tools capable of evaluating other "dimensions", like society and economy (Hofstetter, 1998; Hofstetter, Bare, Hammitt, Murphy, & Rice, 2002; Wrisberg, et al., 2002). Technological decisions are largely made by the private sector, to which economy is the most important factor. Such decisions can lead to unwanted or unforeseen negative societal and environmental ramifications. On the contrary, however, given an unfavorable investment environment, the desired positive societal and environmental benefits of new technologies may never be realized if the decision were to remain entirely with the private sector and the "invisible hand" of free markets. Thus, there are clearly societal and environmental motives for understanding the economic feasibility of a project incorporating preferred technologies, particularly if that technology is novel and undemonstrated at the commercial scale.

Of the factors affecting a private investment decision in a capitalistic economic system, earning a handsome profit is one of the most important (M. S. Peters, Timmerhaus, & West, 2003). A number of methods for calculating profitability are employed in economic analysis, and generally these can be distinguished between those which consider the time value of money and those which do not. For most financial decisions, however, costs and benefits occur at different points in time, and considering the time value of money is essential in understanding the earning power of a large, upfront investment decision (Berk & DeMarzo, 2007). Such methods include Net Present Value (NPV) and Internal Rate of Return (IRR) analysis (also referred to as Discounted Cash Flow Rate of Return (DCFROR) analysis), and both account for the earning power of invested money by employing discounting techniques. In NPV analysis, the NPV is determined by subtracting the present value of all capital investments from the present value of all cash flows. The IRR is the rate of return obtained from an investment in which all investments and cash flows are discounted. It is essentially the discount rate that would give a project a NPV of zero.

1.4.5.1 NPV and IRR Weaknesses

Use of NPV and IRR analysis tools for understanding the profitability of an investment decision – while practical – requires access to detailed cost estimates and other technical data, which is often not available to the public, especially for novel energy process technologies.

1.4.6 General Tool Combination Theory

Tools of any dimension (i.e., environmental, economic, social) can be combined to estimate the impact of a change in the direction toward sustainable development. Wrisberg et al. (2002) nicely illustrate three distinct ways in which this is possible:

- *Overlapping*

- *Consecutive*

- *Parallel*

An *overlapping* use of tools is appropriate when the system definition[2] or mode[3] of analysis is the same, and the overlap may be total or partial depending on whether the same interventions are considered. For example, one may wish to perform a hybrid IO-LCA by combining the individual frameworks (due to the aforementioned benefits) to assess the impacts of a single "unit" product (i.e., 1 MJ of forest-biofuel). The tools in this case overlap because their modes and system definitions are the same, as shown in Figure 1.
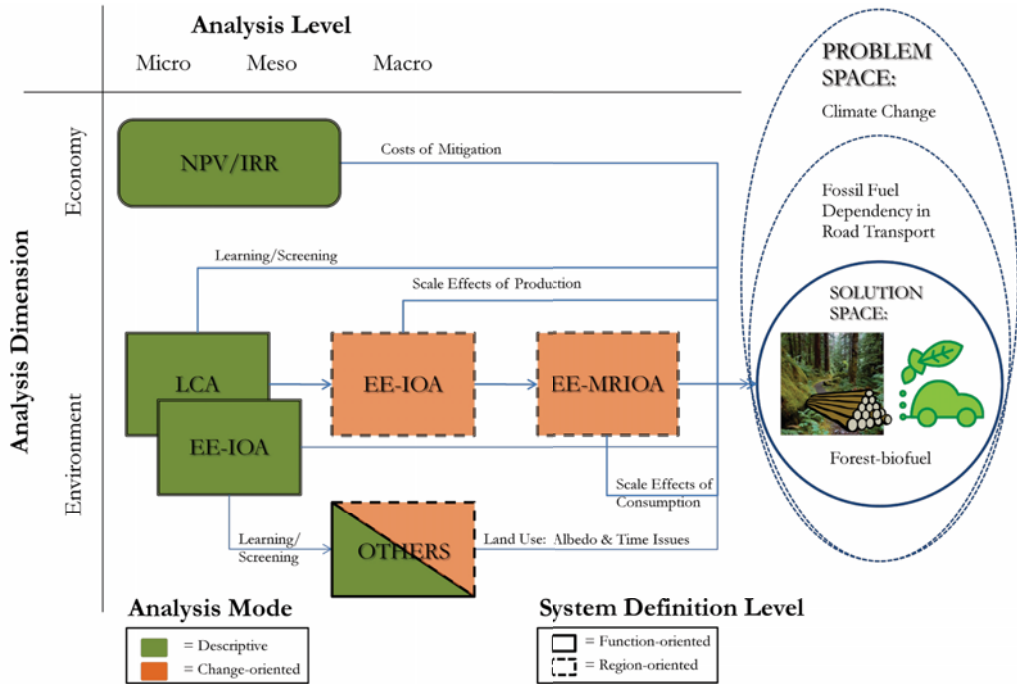
Figure 1. Schematic representation of analytical tool application in this thesis. The blue arrows convey the manner in which a tool is applied to address the main research question (listed over arrow). Positioning in the vertical plane indicates the degree of tool overlap, and positioning in the horizontal plane indicates the degree of tool successiveness. Tools are also positioned to illustrate their scope in terms of "analysis level" and "analysis dimension", adapted from Hofstetter (1998; 2002).

Tools can also be applied *consecutively* where the result from the use of one tool is an input to the use of a second tool, and is appropriate when needing to provide answers to different questions which build on each other. Wrisberg et al. (2002) distinguish between "linear", where the object of the analysis is the same, or "additional", where the object of analysis is different. This is

---

[2] "System definition" tells one which processes are included in the analysis. Wrisberg et al. (2002) use the term "system definition" to distinguish between two types of systems – region- vs. function-oriented.

[3] "Mode" of analysis tells one how the analysis is to be performed, and can be interpreted in various ways. Udo de Haes et al. (2000) distinguish between "full mode" and "attribution mode", whereas Frischknect (1998) refers to mode as either being "descriptive" (describing a system as it is) or "change-oriented" (describing the effects of changes resulting from decisions). Use of the latter terminology is adopted in Figure 1.

practical when decisions require a broad analysis of the question as well as an analysis of specific aspects. An example here might be to apply LCA for the analysis of a single product to screen products and then integrate the process inventory of a preferred product into a region-oriented EE-IO framework for assessing the consequences of change due to a decision at the macro level[4] when the product's production and consumption is scaled up.

Tools applied in *parallel* highlight different aspects of the same or slightly different question. Additional tools may be used to address the same object but different dimension. For example, one might employ various economic tools alongside unit-based LCA to put a cost on pollution abatement. Another possibility is that a decision can be subjected to a number of related but different questions concerning the same dimension but requiring different tools.

1.5 Scenario Analysis

Scenarios of the future are relevant elements in tools applied in Environmental Systems Analysis since they are often used to understand or describe impacts in the future (Hojer et al., 2008). Börjeson et al. (2006) suggest a typology based on the types of questions that are addressed, whether the goal is to predict "what will" happen, explore "what can" happen, or understand "how" a specific target can be met. The usefulness of creating scenarios regarding "what can" happen is to explore situations or developments that are regarded as plausible – so-called *explorative* scenarios (Börjeson, et al., 2006). They are useful in cases when one has a good understanding of the functioning of the present system but is interested in exploring the consequences of alternative developments. They are particularly useful in the case of strategic developments, and are used extensively throughout the thesis.

1.6 Resource Assessment Framework

As previously mentioned, before one is able to make sound technology decisions about biofuels, it makes sense to think about the raw material production potential including the land use requirements first. It is important to ensure that use of the forest resource and the associated land footprint will not pose a serious risk of permanent carbon stock depletion and other adverse environmental and social impact due to market-mediated displacement effects driving global land use change. A guiding resource assessment framework principle is embraced in the thesis to ensure that these risks are minimized: traditional industry demands for forest biomass resources are acknowledged and given priority prior to assessments of the supply originating from forests and wood processing industries (in the form of by-products/residuals), as presented in Figure 2 (i.e., "(B)"). As such, the potential for trade imbalances and other market mediated displacement effects as a result of direct competition for the resource are implicitly minimized following this framework.

---

[4] "Analysis level" describes the positioning of comparative analysis tools in the decision making process in terms of scope. Hofstetter (1998; 2002) has characterized them according to different levels of decision-making: tools may be developed for modeling applications serving to support decisions on the "micro" level (i.e., products, services, or production plants), on the "meso" level (i.e., projects, technologies, or individual sectors) or on the "macro" level (national policy/regulation), where whole countries or the whole world is modeled.
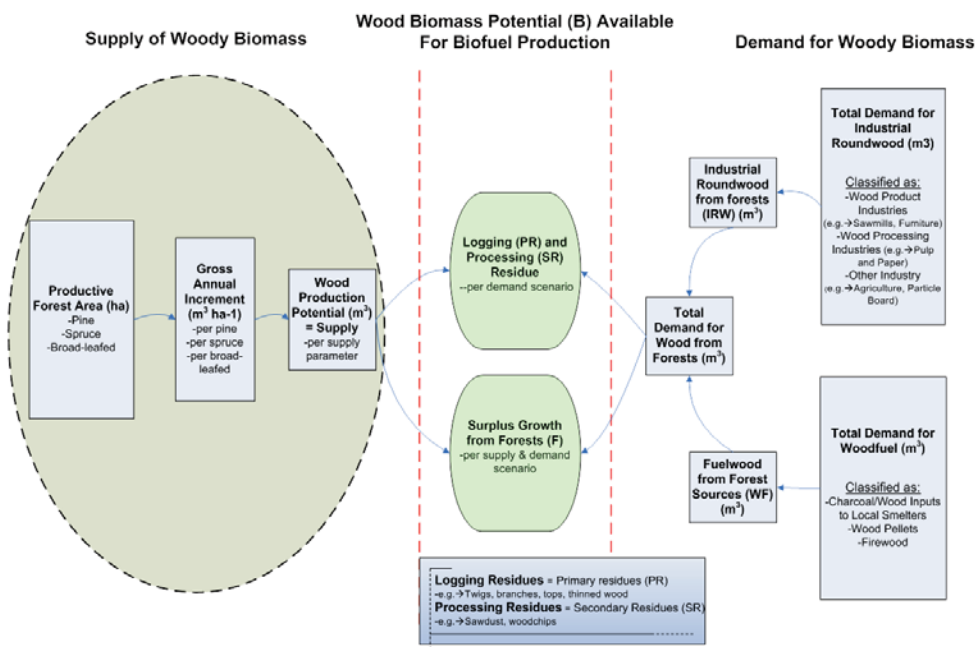
Figure 2. Resource assessment framework used in the thesis, adapted and modified from Smeets and Faiij (2007). For Norway, an economic supply potential parameter is applied to estimate the supply from annual increment which excludes non-profitable increment assuming 1999 market prices.

Throughout the thesis, the term "sustainable" refers to this potential, or the "Wood Biomass Potential (B) Available for Biofuel Production," shown in Figure 2.

1.7 Structure of Thesis

The remainder of the thesis is divided into six parts:

Chapter 2 – Understanding key system facets of environmentally-effective road transportation based on boreal forest-biofuel

In this chapter, the first primary research question is addressed: *How does liquid biofuel produced from boreal forest biomass measure up to fossil fuel in the provisioning of a road transport service, and would any climate gains come at the expense of other environmental or human health impacts?* Life cycle assessment (LCA) is employed to provide a comparative overview of the environmental performance of personal mobility fueled by four alternative E85 product systems, benchmarking results to a gasoline reference system. A case study region in central Norway is chosen, and life cycle inventory analysis is used as a tool to systematically quantify material/energy inputs to, and waste/emission outputs from, systems producing both bio-ethanol and personal vehicles. Life cycle impact assessment is used to quantify the direct, upstream, and downstream environmental risks associated with the complete system delivering the transport service. Contribution analysis is applied to identify aspects of the system – such as key processes and interventions – which are critical to the overall environmental performance. A regional resource assessment is performed so that sustainable volumes of gasoline substitution can be approximated for the case study region.

Novel contributions in Chapter 2

Knowledge of the key environmental hotspots and design aspects of the forest-biofuel product system is generated. Data are compiled on the resource potential of the case study region, and comprehensive life cycle inventories are established for Norway. This knowledge is used as a foundation for addressing additional research questions throughout the remainder of the thesis.

Chapter 3 – Gaining insights into scale-effects of biofuel production and the potential for avoided greenhouse gas emissions from fossil fuel substitution

In this chapter, the second primary research is addressed: *What are the global climate and resource use implications when production of forest biofuel is scaled up commercially in a regional context?* The life cycle inventory of the best performing ethanol product system determined in Chapter 2 is adapted to form an inventory for a renewable diesel fuel product, and the same resource assessment framework of Chapter 2 is applied to quantify the sustainable feedstock potential for Norway. Both life cycle inventories are merged to form a biofuel "sector" which is integrated into a two region hybrid IO-LCA framework to explore scenarios of future biofuel infusion in the Norwegian economy up to 2050. The resulting direct and indirect global climate impacts due to Norwegian biofuel production and consumption are quantified for two fossil fuel substitution scenarios designed around policy targets. A rationale for the approach is provided, and methodological uncertainties and shortcomings are elaborated.

Novel contributions in Chapter 3

The region-oriented systems approach has not been taken previously to evaluate the climate consequences of a commercial forest biofuels industry in Norway. The consumption-based, bi-regional perspective enables a more complete representation of changes in global emission due to scaled forest-biofuel production induced by Norwegian final demand (consumption).

Chapter 4 – From lab to market: The economics of pioneering technology deployment

In this chapter, the third primary research questions is addressed: *What are the costs, both to the public and private sectors, of deploying novel forest biofuel production technologies in the near term?* The perspective of the private investor is taken to understand investment risks affiliated with 1st-of-a-kind commercial biofuel plants in Norway. This is needed to assess the cost-effectiveness of various support policies that might be required by the Norwegian government (i.e., "public") for spurring investment into desirable forest-biofuel technologies, like synthetic diesels, for the purpose of more immediate climate change mitigation. Key findings are discussed and uncertainties and limitations are elaborated.

Novel contributions in Chapter 4

Technical risk inherent to novel energy process technologies operating at the commercial scale is quantified using statistical models of cost growth and plant underperformance. This results in a more conservative production cost estimate of forest-biofuel than what is typically reported in

the literature. This cost estimate is then used in policy analysis for identifying trade-offs between alternative financial support strategies when both government and private sector cost/profit standards are considered simultaneously. The public costs of GHG-abatement linked to the support policies are assessed in light of a range of uncertain future oil prices.

Chapter 5 – Silver bullet or buckshot? Understanding forest biofuel's niche in regional climate friendly road transportation

In this chapter, the fourth primary research is addressed: *How would other factors beyond lowered carbon intensity of the fuel supply affect resource use and climate impacts of road transportation throughout the region?* In Chapter 3, the climate implications related to the scale effects of forest biofuel production in the Norwegian economy is assessed with a focus on the carbon intensity of liquid fuels consumed in key fossil-dependent sectors. In this chapter, other aspects contributing to GHG emission growth in road transport are explored, such as fuel intensity and overall demands for road transport activity. The geographic scope expands to include Sweden and Finland, and a more detailed representation of regional road transport consumption and forest biofuel production is considered in a scenario-driven assessment. A rationale for the approach is provided, and methodological uncertainties and modeling limitations are elaborated.

Novel contributions in Chapter 5

The IPAT analogy is used in the design of scenario parameters for assessing the degree of absolute decoupling between growth in the demand for road-based transport and GHG emission. Structural aspects of road transport consumption are partly used to define consumption parameter variables, and technology scenarios regarding regional fuel mixes and energy use efficiencies are created and modeled in a mixed-unit, multi-regional input-output framework that is inclusive of global trade.

Chapter 6 – The role of the Scandinavian boreal forest in climate protection

The fifth and final primary research question of the thesis is addressed in this chapter: *What are the climate implications attributed to the use of forest biofuel when forest carbon cycle dynamics and biogeophysical land use effects like surface albedo changes are considered in climate impact analysis?* Full accounting of effective carbon sinks and sources are considered in a new region-oriented scenario-based study of forest-biofuel in Norway. The study goes beyond the carbon cycle and approximates the magnitude of the climate perturbation attributed to albedo changes in forests when forests are managed more intensively for biofuels. A change-oriented, or marginal perspective, is adopted for climate impact assessment.

Novel contributions in Chapter 6

Forest carbon cycle modeling is combined with life cycle inventory modeling and radiative forcing impact analysis to quantify climate impacts/benefits in Norway associated with forest biofuel production and consumption at the national level. For the first time, climate impacts

from albedo changes are quantified and linked explicitly to the combined forest management plus biofuel system, and in addition, an explicit representation of time is included in impact assessment.

Chatper 7 – Summary and Outlook

The main academic contributions of the PhD thesis are summarized, and a direction for further research is outlined.

Chapter 2: Understanding Key System Facets of Environmentally-effective Road Transportation based on Boreal Forest Biofuel

2.1 Questions

Climate change mitigation and security of energy supply are the omnipresent socio-political motives from which the academic research stems, but the extent to which a future transportation system based on biofuel will allay these concerns without adding to unwanted burdens in other areas, such as, for example, increased damage to human health or to local air and water quality – will remain unanswered until the following research questions are adequately and systematically addressed:

- Given a defined region, what might a typical alternative transportation system operating on forest-derived biofuel look like in terms of the major processes of the supply chain(s) that comprise it?

- In the provisioning of a single unit of transport service afforded by forest biofuel as an energy carrier, what might the typical material/energy inputs and waste/emission outputs of these processes be, and of the system as a whole *ceteris paribus*?

- What are the direct, upstream, and downstream environmental risks associated with these?

- Which aspects of the system are critical to the overall environmental performance, and how would it measure up to today's system providing a comparable service?

- Given a constraint in the resource potential in which the system operates, what are the scalability implications of the alternate technology?

2.2 Applicability of LCA

The term "environmentally-effective" is obviously value-laden and depends on the specific societal objective and alternative system being compared. If the goal is to reduce the risk of climate impact while avoiding increased impacts in other impact categories relative to that imposed by the current technological system, forest-derived biofuel could play a role at being "environmentally-effective", and various assessment tools can help objectively quantify that role. Currently, a wide range of environmental assessment tools have been used in comparative analysis and/or are designed to quantitatively evaluate some of the risks identified with new technologies, like biofuel. For those designed for use as decision support rather than monitoring, some are less geared for comparative purposes than others, such as, for example, Substance or Material Flow Analysis (S/MFA), Environmental Impact Assessment (EIA), and Environmental Risk Assessment (ERA). ERA has proven to be a powerful environmental management tool in the assessment of risk of specific chemicals in the environment (Owens, 1997a). Like S/MFA and EIA, ERA functions, however, as an absolute assessment tool and thus requires very specific and detailed spatial and temporal information, particularly with respect to human and environmental exposure conditions (Olsen et al., 2001). Those which are more geared for decision-support include tools such as Life Cycle Assessment (LCA), Risk Tradeoff Analysis (RTA), Comparative Risk Assessment of Alternatives (CRAoA), Programmatic Comparative Risk

Assessment (PCRA), and Cost-Benefit Analysis (CBA), for example, and can be characterized with respect to their coverage of different types of risk, the scope of the analysis level and assessment dimensions, the type of decision making principle, and the ways in which distributional questions are dealt with (Hofstetter, et al., 2002; Pearce, Atkinson, & Mourato, 2006). Hofstetter et al. (2002) formulate five pertinent questions that the analyst or decision maker should answer in the tool selection process. Regarding the analysis level and dimensional scope, is the focus on environmental impacts or will societal and economic effects be assessed, and at what decision making level (micro/product to macro/national policy)? Regarding the decision-making principle, is it about prioritizing measures, selecting a single least damaging alternative, or about implementing any measure with net benefit? Regarding distributional aspects, do they need to be considered, and if so, how much of a reduction in population risk should be sacrificed at the expense of equity in distribution? What are the types of risks that need to be covered, and to what extent?

The specific research questions postulated in Chapter 2.1 provide the necessary context for answering these questions for choosing the most appropriate decision-analytic framework. Regarding the scope, since the research objective is to evaluate environmental dimensions of change at the unit-product analysis level (single service or good), LCA is the most applicable assessment tool (Hofstetter, 1998) relative to the others mentioned above. Additionally, since direct, upstream, and downstream risks are the requisite risk coverage, LCA is well-suited here as well. CBA has similar coverage but is limited by the inclusion of endpoints that can be monetized (Guinée, et al., 2002; Hofstetter, et al., 2002; Pearce, et al., 2006). Further, because the decision-making principle is based on the minimization of environmental impact, and because distributional issues are not of concern, LCA is also appropriate relative to some of the other tools like RTA, for example.

It is therefore determined that LCA is the most applicable decision-analytic framework to address all questions posed in Chapter 2.1 in a systematic and structured manner. LCA can assist decision makers to ensure that the proposed technological solutions thought to mitigate environmental problems of one type do not impose additional problems of another. There are a range of various decision-making situations in which LCA studies can be performed, ranging from company internal to public comparative use (Guinée, et al., 2002). These different situations can impose different requirements on the type of decision procedure which has to be followed. For example, in the context of environmental assessment of biofuel, the use of LCA to directly support policy making requires standardization to reduce the unpredictability of outcomes arising from inconsistent choices in allocation, system boundaries, and treatment of biogenic CO_2 (Cherubini & Strømman, 2010; van der Voet, Lifset, & Luo, 2010). The extent to which procedure adheres to strict methodological standards depends on the situation, where, for example, in a situation of global exploration or company-internal innovation, there is generally less need for strict process regulation than in a situation involving disclosure or justification to the public (Guinée, et al., 2002).

LCA may also be useful when applied in non-decision making situations, such as the identification of improvement possibilities or choice of environmental performance indicators and market claims (ISO 14040:2006). Simply put, LCA can be a practical learning tool, applied to explore the environmental properties of the product system under study and to acquire insights

into the relationships of the production system (Baumann, 1998). In the context of biofuel assessment, when used as a learning or an informative tool, LCA can help identify important improvement options in the alternative system without needing to make difficult decisions regarding allocation, for example, and in this context is usually more robust than comparative applications (van der Voet, et al., 2010).

2.2.1 Attributional LCA

In general, all LCA applications aim at some form of change or improvement, some in more direct ways (i.e., decision making), and some in more indirect ways through influencing market behavior or through identification of improvement possibilities (Baumann & Tillman, 2004). Generally, when the research objective is to describe a system's attributes using average input-output data that best typifies that system's performance, attributional- or descriptive-LCA is best-suited. In other words, the goal of attributional-type LCA is to describe the environmentally relevant physical flows to and from a product (or service) system under the assumption of *ceteris paribus*, relying on average data representative of average environmental performance for the provisioning of that product (or service). Alternatively, when the research objective is to describe how environmentally-relevant flows will change in response to a possible decision, employing a consequential- or change-oriented-type LCA may be more applicable (Curran, Mann, & Norris, 2005). In consequential- or change-oriented LCA, the system's boundaries are typically defined to include the activities contributing to the environmental consequence of the change irrespective of whether or not the change occurs within or outside the system under study, and as a result, the process of system expansion becomes an inherent part of the LCA study (Ekvall & Weidema, 2004; Finnveden et al., 2009). In other words, system expansion inherently measures the net system change. This type of LCA employs marginal data representing the effects of a small change associated with the provisioning of the system's functional unit. The effects of the change largely depend on economic models, thus consequential-type LCAs usually embrace additional economic concepts like marginal production costs, elasticity of supply and demand, and dynamic models. Because of this, it is conceptually more complex, and the results obtained are highly sensitive to the assumptions made (Finnveden et al, 2009) therefore introducing limitations concerning accuracy. Further limitations of consequential LCA concern relevance: certain decision makers can be more interested in knowledge of the environmental properties of systems rather than knowledge of the effects of changes within the life cycle (Ekvall, 2002).

Both attributional- and consequential-LCA can and have been applied for modeling future systems (Ekvall, Tillman, & Molander, 2005; Sandén & Karlström, 2007). However, the goal of applying LCA – as defined by the specific set of research questions outlined in Chapter 2.1 – is to assess the environment burden of the prospective technological system assuming a status-quo situation in order to learn about the new system for identifying improvement possibilities. This requires one to describe the performance of system and its attributes as wholly and completely as possible, thus attributional-type LCA is applied in the first case study.

2.3 Global Warming Impact Paradigms in Biofuel LCA

LCA models are inherently static; that is, they provide a "snapshot" of environmental impact where the snapshot is based on all that occurred over the time interval of the snapshot. In other words, LCA essentially integrates over time, and all impacts, irrespective of the moment that they occur, are equally included (Udo de Haes et al., 1999). This means that, in the case of biofuel, the

"cooling" impact that occurs over the biomass growth time period (due to the removal of CO_2 from the atmosphere) is usually not represented. Thus, to compensate, the "warming" impact that occurs once biofuel is combusted is often neglected because it is assumed the quantity assimilated during growth will approximately equal that which is released upon being oxidized ("carbon neutrality" principle). This practice is so widespread in biofuel LCA application that, out of 67 studies[5] evaluated in a recent biofuel LCA review study performed by van der Voet et al. (2010), 63 failed to even state this assumption. A less widespread convention in LCA of bioenergy (2 out of 67 in (van der Voet, et al., 2010)) has been to explicitly account for the biogenic CO_2 intervention at each stage; in other words, as negative emissions during biomass growth and positive emissions during biomass combustion – an inventory modeling approach adopted by some circles (Ecoinvent, 2009; Rabl et al., 2007). The problem with this approach is that it complicates allocation, which may put credits for extracted CO_2 in a different part of the multiproduct chain than the debits for emitted CO_2 – while ignoring biogenic CO_2 altogether would not have this effect (Luo, van der Voet, Huppes, & Udo de Haes, 2009). For example, for a biofuel product derived from waste biomass, the CO_2 emission credits would have occurred upstream outside the system boundaries and the full global warming impact would be attributed to the biofuel.

It would appear that the temporal issue of biogenic CO_2 impact has been sufficiently dealt with by researchers in other disciplines, but thus far a proper representation of time in global warming impact assessment of biogenic CO_2 for use in LCA has been missing. A recent solution proposed by Cherubini et al. (2011) is a characterization model which incorporates the effective sink capacity of the biomass feedstock which explicitly expresses the warming impact as a function of sink re-growth rate and rotation time, a so-called "GWP_{bio}" characterization factor. While intuitive, use of such a metric requires an assumption that the same biomass species used as feedstock for biofuel is immediately replanted following biofuel combustion having the same growth rate and rotation period. Nevertheless, such a metric is needed in unit-based life cycle assessments of biofuels.

2.3 Introduction to Case Study

The technologies to produce biofuel from forest biomass are immature, thus a future-oriented LCA case study is performed with the research motive stemming from the need to identify important improvement options in the product system before strategic technological decisions are implemented at the societal level. One biofuel product, two conversion technologies producing at a single scale, and two logistical alternatives upstream in the biomass supply chain are considered. An additional objective of the case study is to assess the biomass resource potential in the case region to quantify the fossil substitution potentials and approximate the degree of regional self-sufficiency. In keeping with conventional LCA practice, a characterization factor of zero is applied to biogenic CO_2 emission.

---

[5] Including Paper I

Life Cycle Assessment of Second Generation Bioethanols Produced from Scandinavian Boreal Forest Resources: A Regional Analysis for Middle Norway

2.5 Uncertainties and Limitations

2.5.1 LCI

Data variability and uncertainty are inherent aspects of LCA as with many decision support tools. Different types of uncertainty have been described using different typologies, although most can be classified as one of three types: parameter, scenario, and model uncertainty (Huijbregts, Gilijamse, Ragas, & Reijnders, 2003; Lloyd & Ries, 2007). The largest uncertainty aspect of the performed case study is of course related to a lack of knowledge regarding the future forest-biofuel system. To this end, measures to estimate parameter uncertainty associated with the life cycle inventory data via statistical methods could not be justified. Instead, parameter and data quality uncertainty regarding process inputs, emissions, and technology characteristics were addressed qualitatively. Scenario uncertainties regarding allocation procedures, time scale, geographic scale, and choice in functional unit were mostly addressed qualitatively. To some extent geographic scale issues were dealt with quantitatively through sensitivity scenarios to evaluate the influence of changes in transportation parameters upstream in the biomass supply chain.

2.5.2 LCIA

Model uncertainty in the LCA case study stems from the choice in characterization factors applied in impact assessment. The midpoint impact assessment performed was limited to four impact categories. State-of-the-art midpoint impact assessment methodologies were applied at the commencement of the study (Leiden University, 2001). For acidification and eutrophication impacts, the applied characterization factors were derived from a site generic characterization model in which the receiving environments were excluded. However, acidification and eutrophication impacts are often local or regional in nature, and the assumption of a standard global homogenous receiving environment can disregard large and unknown variations in the actual exposure of the sensitive parts of the environment (Finnveden, et al., 2009). The location of an emission source can lead to different responses in the surrounding environment, depending on ecosystem sensitivities and local atmospheric conditions (Bare, Norris, Pennington, & McKone, 2003; Hettelingh, Posch, De Smet, & Downing, 1995; Huijbregts, Schöpp, Verkuijlen, Heijungs, & Reijnders, 2000; Posch, Hettelingh, & De Smet, 2001; Potting & Hauschild, 1997). Application of spatially-explicit characterization factors for both acidification and terrestrial eutrophication for Norway, for example, may have led to different conclusions (Huijbregts, et al., 2000; Seppälä, Posch, Johansson, & Hettelingh, 2006). However, spatial differentiation in LCIA can increase the complexity of LCA, requiring more information about the specific location of emission sources, particularly if a significant source originates in the background system. This can be overcome, however, with steps towards database-wide regionalization that couple existing regionalized characterization factors with large life cycle inventory databases like Ecoinvent (Mutel & Hellweg, 2009).

While direct land use change (dLUC) and the impacts thereof are not a concern when sourcing biomass from existing productive forest areas, more intensive land use via occupation does pose an increased threat to biodiversity, lowered biotic productivity, and reduced soil quality (Lindeijer, Müller-Wenk, & Steen, 2002; Mila i Canals et al., 2007; Udo de Haes, 2006). While several recent publications have proposed methods on how to include these impacts (Koellner & Scholz, 2007,

2008; Michelsen, 2008), there is currently no agreement of how these impacts should be included in LCA (Finnveden, et al., 2009).

2.6 Summary and Contributions

As a decision-analytic tool, LCA has provided a quantitative background for comparing the environmental risks of several forest-biofuel product systems against each other and against a fossil reference system. As an informative and learning tool, LCA has provided a structured and systematic foundation leading to an increased understanding of key system facets and parameters influencing the environmental performance of forest-based biofuel transport. In the biofuel production system, minimization of wood chip storage and biomass transport upstream in the supply chain will be one important criterion to strive for in the regional planning of future forest-biofuel supply chains. Co-location of biofuel production facilities with industrial suppliers of residual biomass will be an environmentally-effective (and likely cost-effective) strategy. The most vital component influencing the environmental performance of the biofuel production system is the biomass-to-biofuel conversion efficiency. Decreases in this parameter correlate with increases in activity upstream in the supply chain, leading to significant increases in environmental impact and use of resources.

LCA results have also indicated that achieving a reduction in environmental impact from road transport based on forest biofuel will also require improvement measures in the vehicle production system, as the production and operation efficiency of the vehicle itself contributes significantly to the overall system performance. Extending the lifetime and durability of passenger vehicles in addition to improving fuel efficiency (via weight reductions, engine size reductions, etc.) are key improvement areas that warrant inclusion in future transport policy.

2.6.1 Research Implications

From an environmental impact and resource use perspective, findings of the LCA case study demonstrate the superiority of thermochemical over biochemical production technologies in stand alone, centralized, large-scale applications based on lignocellulosic feedstocks. To this extent, other liquid biofuel products produced thermochemically ought to be investigated. Additionally, findings support a research need to evaluate the environmental implications when the system boundaries are expanded and biofuel production is scaled commercially, that is, when produced at the regional or national level.

Chapter 3: Gaining Insights into Scale-effects of Biofuel Production and the Potential for Avoided Greenhouse Gas Emission from Fossil Fuel

3.1 Questions

In Chapter 2, using Norway as the case study region, knowledge of the technical boreal forest-biofuel product system was acquired which led to the development of comprehensive life cycle inventories for centralized (large-scale) bio-ethanol production. Results of the life cycle impact assessment for the alternative product systems revealed forest-biofuel's potential to reduce global warming impact without significantly imposing tradeoffs with human health and other environmental impact categories. However, due to the scope and boundary conditions of the function-oriented assessment, potential indirect effects attributed to scaling up biofuel's production and use cannot be measured when taken out of context of the larger, economy-wide production system that produces multiple goods and services simultaneously. Therefore, the following detailed research questions are postulated:

- In the event that a national policy were to be implemented in Norway having biofuel targets similar to those outlined under current EU legislation, what are the potential climate implications of commercial forest biofuel production and use at the national level – both in the near and over the longer-term time horizons?

- What is the approximate sustainable domestic forest-derived resource potential in the region?

- If the region were to prioritize use of the resource for the production of forest biofuel, what would be the fossil fuel displacement potential over the near- and medium-time horizons?

- What are the approximate avoided GHG emissions associated with this potential, both domestically and globally?

3.2 The Case for Fischer-Tropsch Diesel

An ethanol-based biofuel product was chosen for the LCA case study in Chapter 2 due largely to the availability of reliable process engineering data stemming from the significant amount of research activity surrounding cellulosic ethanol in Sweden and in the USA at the time. However, ethanol in its neat form cannot directly replace gasoline in today's spark-ignited vehicles without some minor engine and other vehicle modifications. Further, an analysis of vehicle registration trends in Norway over the past decade reveals that diesel-engined vehicles are increasing in market share and have been forecasted to dominate most of the light duty vehicle fleet in the near future (TØI, 2008). Additionally, diesel fuel powers most of today's heavy duty vehicles owed in large part to its superior (relative to gasoline) physical and chemical properties (i.e., higher cetane value, higher energy density) making it an ideal energy carrier in long range and heavy usage applications (compression ignition engines are more durable and reliable). It is likely that heavy duty freight and passenger transport will continue to rely on diesel fuels for some years to come (Ohlrogge, et al., 2009; Savage, 2011). Thus, it makes sense to consider other forest

biofuel products that could pose as suitable near- and medium-term diesel substitution candidates like synthetic diesel derived from Fischer-Tropsch processes.

The Fischer-Tropsch process is a thermochemical process which upgrades short-chain hydrocarbons into long-chain alkanes and waxes using metal catalysts under high heat and pressures. Fischer-Trospch reactors are usually coupled with gasification processes, a process producing high quantities of syngas from solid or gaseous feedstocks like coal, biomass, or natural gas. Syngas is a high concentration mixture of elemental CO and H_2 which is fed into FT reactors serving as the building blocks of synthetic fuels like Fischer-tropsch diesel (FTD).

At the commencement of the thesis work, little process engineering (mass and energy balance) data surrounding wood chip-to-FTD production data existed in the public sphere, particularly for large-scale, autonomous (energy self-sufficient) processes like the thermochemical ethanol process of Chapter 2. It was therefore chosen to adapt the thermochemical ethanol life cycle inventory of Chapter 2 to wood chip-to-FTD yield and conversion efficiency information from the literature for use in the thesis.

3.3 Technology Scenarios and the Applicability of EE-IOA

The application of input-output models to analyze scenarios about the future can be instrumental in the exploration of impacts of technological or policy change (Faber, Idenburg, & Wilting, 2007; Höjer et al., 2008; Tukker, Eder, & Suh, 2006; Harry C. Wilting, Faber, & Idenburg, 2008). This is owed to its region-oriented system definition (i.e., inclusive of all technological interactions and production activities of an economy) and macro-level decision analysis scope (applied to evaluate the consequences of national policy decisions). By allowing transparent dissociation of the economic and environmental dimensions of development, an EE-IO model can provide a good understanding of the types and magnitudes of impact biofuels may have when scaled commercially. They can be applied in a straightforward manner to evaluate future policy scenarios and to gain valuable insight into long-term sustainability of technological choices like second generation biofuels produced from forest biomass.

Top-down economic-market-based models like computable general equilibrium (CGE) models can also provide a consistent macroeconomic framework within which energy-economy-environment interactions can be examined. However, because CGEs are designed to take into account of a multitude of interactions among sectors of an economy, they cannot simultaneously incorporate detailed specifications of particular technologies and devices that underlie the sector-level projections (Schafer & Jacoby, 2003). Technical change at the sectoral level is often an artifact of the aggregation level chosen (Jacoby, Reilly, McFarland, & Paltsev, 2006), and as a result, isolating and identifying explicit cause-effect relationships linking a specific technology to a specific quantity of environmental impact proves difficult in "top-down", CGE-type modeling. This can be overcome in hybrid IO-LCA type frameworks that incorporate detailed technology data together with technology-specific environmental externality data, both derived with assistance from "bottom-up" process engineering models. Thus impacts can more easily be measured, observed, and attributed to specific technologies.
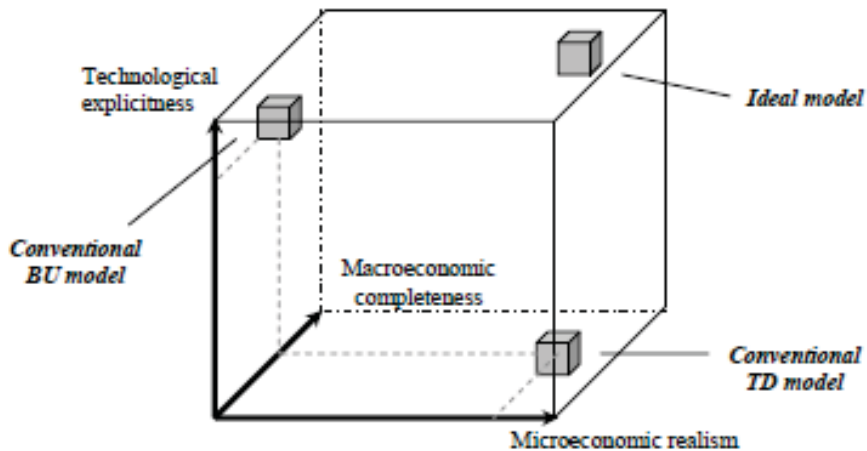
Figure 3. Representation of general strengths and weaknesses of "Bottom-up" (BU) and "Top-down" (TD) energy-environment-economy models, where technological explicitness is traded for microeconomic realism and vice versa. Figure source: (Hourcade, Jaccard, Bataille, & Ghersi, 2006).

However, this is not to say "top-down" models like CGEs are not needed. Top-down CGEs generate economic trajectories which present a strong internal consistency between economic and energy trends (McFarland, Reilly, & Herzog, 2004). They often portray a better dynamic representation of macroeconomic feedbacks and more realistic interactions at the microeconomic level which can serve to inform important long-term structural changes. Results of top-down studies can be used to improve the robustness of scenarios created for analysis in static, hybrid IO-LCA frameworks – such as adjustments to input structures and final demands for energy products and services over time, for example.

3.4 The Integrated Hybrid LCA-IO Framework

As touched upon briefly in Chapter 1.4.4, the benefit of any hybrid LCA framework stems from the fact that both LCA's and IOA's strengths are exploited. In other words, sector aggregation contributing to uncertainty in the analysis of products in IOA can be reduced by incorporating detailed bottom-up technology data unique to single products or production processes (i.e., by combining with process-based life cycle inventories); similarly, truncation error associated with system boundary incompleteness of process LCA can be reduced by coupling with economic input-output data to capture the full activity occurring upstream in the supply chain. The practice of combining an LCA of a single product with an input-output model of an entire economy or region to capture indirect impacts that would otherwise be ignored has gained favor in recent years (Strømman, Hertwich, & Duchin, 2009).

Three types of "hybrid-LCA" methods are predominantly applied in environmental systems analysis, with the differences stemming from how the process LCI data and economic IO data are integrated computationally (Suh & Huppes, 2009; Suh et al., 2004). They are referred to as *Tiered Hybrid Analysis*, *IO-Based Hybrid Analysis*, and *Integrated Hybrid Analysis*, shown in Figure 4 (Suh & Huppes, 2005, 2009).
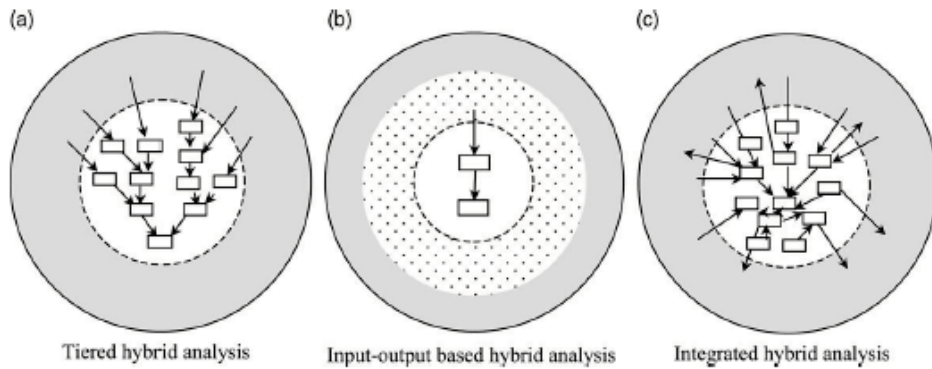
Fig. 3. Interactions between process-based system and IO-based system of hybrid analyses.

Figure 4. Conceptual illustration of the three types of hybrid-LCA frameworks. The solid outer line indicates the overall system boundary, and the dashed line represents the boundary between the process-based system (solid white area) and the IO-based system (solid gray area). In *Tiered* frameworks (a), the process inventory is augmented with IO data. In *IO-Based* frameworks (b), industry sectors can be disaggregated, represented by the dotted white area. In *Integrated* frameworks (c), the process system is fully embedded and linked with the IO system, represented by arrows shown in both directions. Figure source: (Suh & Huppes, 2005).

The three frameworks vary with respect to geographical system boundary, technological system boundary, data requirements, time- and labor-intensiveness, data uncertainty, and simplicity of application. The choice of method depends on the research question. The "Integrated Hybrid" IO-LCA framework is appropriate when the product system under focus is produced in such high volume that it is not negligible with respect to the rest of the economy, making it the preferred choice for addressing questions about the consequentes associated with the up-scaling of new technologies and products, as the indirect environmental and economic effects linked to demands imposed by other industrial sectors and households can be sufficiently quantified.

In case of scaled production of forest-biofuel, where the scope of decision analysis lay at the "meso" or "macro" level (Hofstetter, 1998) and the system requires a "region-oriented" definition (as opposed to "function-oriented") (Wrisberg, et al., 2002), the *Integrated* hybrid IO-LCA framework is well-suited. The framework may be applied to assess historic impacts of existing technologies in so-called "descriptive" analysis mode – or in future-oriented studies where the goal is to explore plausible scenarios about new technology infusion or technology change – in so-called "change-oriented" modes of analyses (Frischknecht, 1998).

3.5 Introduction to Paper II

An integrated hybrid model is built and employed to evaluate the future impacts of forest biofuel infusion throughout Norway. Final demands and GDP over time are derived from top-down energy-environment-economy models and used as scenario parameters to drive the hybrid model. Outputs from partial-equilibrium studies of the European energy sectors are used to adjust transport and power generation sector technical coefficients to account for improvements in energy efficiency over time. Biofuel demand scenarios are linked to policy targets and the resulting biofuel output/fossil displacement scenarios are quantified in light of resource constraints.

Environmental Assessment of Wood-Based Biofuel Production and Consumption Scenarios in Norway

3.7 Uncertainties and Limitations

Apart from the aggregation uncertainty (i.e., input-output and factor data are aggregated over a number of producers within one industry), IO-based environmental assessment models have other inherent sources of uncertainty related to the life cycle inventory data. The IO source data itself may be a source of uncertainty, or the multipliers used to convert monetary values into physical quantities in hybrid analysis may be disproportional across industries (i.e., $5 electricity sold to service sectors may not be proportional to $5 sold to manufacturing sectors in physical quantities). Additional uncertainty can stem from the homogeneity assumption, that is, the assumption that each industry class produces only one type of commodity which can lead to a misallocation of production factors. Another source of uncertainty is import assumption uncertainty – the assumption that the foreign industries supplying the imports have the same factor intensities identical to the domestic industries. This uncertainty is particularly relevant for singe region IO-models and when the economy of the focal region is open with respect to trade.

In Paper II, disaggregation of the Norwegian forestry sector – the dominant auxiliary sector to biofuel production – was performed to reduce IO-data aggregation uncertainty. Allocation uncertainty was not addressed, but is likely to have little impact on results since the output of the forestry sector is relatively homogenous. The largest uncertainty of the modeling framework chosen for analysis is import assumption uncertainty, as Norway is an open country reliant on many imports. This was largely overcome by linking the Norwegian IO tables with 15 regions of the EU – Norway's largest supplier of imports in the corresponding data year.

3.8 Summary and Conclusions

It should be noted that quantitative uncertainty analysis is rarely attached to IO-based LCA because often basic uncertainty information for individual elements of an IO table is generally unavailable (Suh & Nakamura, 2007). Nevertheless, the measures described above likely minimized the larger sources of uncertainty stemming from the modeling framework, and the results may be seen as robust with respect to the magnitude of avoided fossil fuel consumption and global GHG emission reductions when direct and indirect effects within Norway and its major trade partner are considered.

Quantification of the domestic resource base in Norway allowed for exploration of scenarios of forest-biofuel production and consumption over the near- and long-term time horizons. Should Norway prioritize fossil fuel displacement in road transport and other sectors heavily dependent on the use of liquid fuels, significant emission reductions could be realized while positioning itself on a path towards long-term energy stability. Such a strategy would serve to mitigate the negative socio-economic effects of a waning petroleum industry, spurring new industry development and contributing to a more resilient and diversified economy over the long term.

3.8.1 Research Implications

Two important implications can be drawn from the study: i) Assuming Norway is to strategically pursue development of a commercial forest-biofuel industry, what are the near-term costs, and what economic support policies might need to be implemented? ii) What is the approximate potential for Nordic Europe as a whole to lower the carbon intensity on road-based transport, and how would the other variables like energy intensity and overall consumption activity shape

the sustainability picture? These main questions are explored in greater detail in Chapters 4 and 5.

Chapter 4: From Lab to Market: The Economics of Pioneering Technology Deployment

4.1 Questions

In Chapter 3, domestic and global climate implications of a commercial forest biofuels industry producing both ethanol and FTD in Norway up to 2050 were assessed. It was concluded that significant reductions in global warming emissions stemming from fossil fuel combustion could be realized should a national policy promoting commercial forest-biofuel production be implemented. Given that such a policy were to be pursued in Norway for these motivations:

- How cost-effective are forest biofuels in the near-term, both to the public and private sector?

- What are the private sector risks, and how might the cost of conventional fossil fuels affect prospects for the development of a commercial forest biofuel industry?

- What types of economic support and financial incentive policies are needed to ensure that costs to the public are minimized and risks to the private sector are reduced and/or compensated?

4.2 Impediments to Early Commercial Experience

As mentioned in Chapter 1, forest or lignocellulosic-based biofuels are not produced commercially in any significant quantity anywhere in the world (IEA, 2010). Realizing national energy security and climate policy targets afforded via forest biofuels in the near-term requires rapid technology deployment and diffusion. Technological learning via cumulative production experience with novel energy technologies has been shown to reduce costs (inverse "S-curve") and accelerate diffusion (Grübler, Nakicenovic, & Victor, 1999a, 1999b). Acquiring cumulative production experience necessitates commercial deployment, and commercial deployment necessitates investment, typically from the private sector. Investment decisions are largely based on anticipated future economic performance of a project. Future performance of projects incorporating novel technologies is often more uncertain than established technologies – an inherent investment risk. Such risk can be reduced or compensated with support from the government but should first be understood so as to minimize too much transfer to the government.

4.2.1 Technical Performance Risks and Uncertainties

The decision to commercialize a new technology depends on a realistic evaluation of its economic viability, and realism calls for reasonably accurate estimates of the capital investment needed to design and construct a plant that will produce the desired product competitively. Earlier research on commercializing first-of-a-kind energy process technologies found repeated failure to anticipate actual costs, and frequent disappointing performance (Merrow, Phillips, & Myers, 1981). Early cost estimates for technically advanced plants are characteristically far below actual costs, and troublesome system performance problems are much more likely for advanced systems than for systems with prior commercial experience. Pioneer forest-biofuel production plants have proven to be no exception (Denver Business Journal, 2011; Deutmeyer, 2010). Risks stemming from technical performance uncertainty leading to capital cost growth and plant

underperformance factor into an investment decision and warrant attention in the economic assessment of advanced biofuel production technologies.

4.2.3 Market Risks and Uncertainties

Market risk is the risk that the value of the investment project will decrease due to changes in the value of the market risk factors – such as stock prices, interest rates, foreign exchange rates, and commodity prices. For investment projects producing liquid fuels as a main product, the most relevant is *commodity risk*, particularly the risk associated with a volatile and uncertain future oil price (Bartis, Camm, & Ortiz, 2008; Camm, Bartis, & Bushman, 2008). With the possibility that oil prices could fall in the near to medium term, the financial risk surrounding the initial biofuel production investment is appreciable, particularly given the large capital investment required for even medium-scaled forest-biofuel plants.

4.3 Applicability of Investment Analysis

For the Norwegian government to encourage the early participation of industry in the forest-biofuel enterprise, technical performance and oil price uncertainty are two important risk factors that must be understood before economic support policy that transfers a portion of these risks to the government can be implemented. Understanding these risk factors, combined with observations from successful voluntary agreements in the commercial world, can lead to the identification of principles that the government can use to design a relationship with a private investor that is likely to ensure that early forest-biofuel production experience occurs cost-effectively. Such a relationship yields investor and government behavior that, in turn, generates a set of cash flows to and from investor and government over time.

4.3.1 Applicability of NPV and IRR Analysis

Detailed analysis of cash flows between government and investor over time provides the means to assess the effects of choosing specific values for the attributes of prospective financial incentive instruments, such as, for example, the level of price support, the size of a tax credit, or the specific terms of any net income-sharing agreement. Net Present Value (NPV) and Internal Rate of Return (IRR) analysis take these cash flows as given and assesses their effects on the investor and the government. It measures effects on an investor in terms of changes in the investor's real (adjusted for inflation) after-tax IRR – a measure of profitability. It measures effects on the government in terms of changes in the real NPV of cash flows to and from the government when assessed at the discount rate set by the Ministry of Finance for investments of this kind. Information about government NPV can be used to compare the cost to the government of increasing the private IRR in different ways afforded by various economic incentive instruments. First, however, it is important to understand how profitability of an investment decision is viewed from the eyes of the private investor. The NPV to a private investor can be expressed as Eq. 1:

$$NPV = \sum_{j=1}^{N} (1+i)^{-j} \left[\left(Q_j * p_j - c_j \right)(1-\phi) + r_j + \phi d_j \right] - \sum_{j=-b}^{0} (1+i)^{-j} T_j$$

(1)

The first summation on the right side of the equation yields the present value of annual cash flows over the operation period N (upon start up), and the right side yields the present value of the capital investments T, with $-b$ being the year in which the first investment is made in the project with respect to zero as the start-up time. Cash flows and investments are discounted a single time at the end of the year j using the discount factor i. The post-tax NPV of the investment is thus the present value of all cash flows less the present value of all investments. Discounted annual post-tax income equals the quantity Q of the product produced times its price p less operating costs c plus the cash recovered from working capital and the sale of physical assets r. Depreciation d is written off as an expense for tax purposes, thus the positive term ϕd results from subtracting depreciation as an expense before income taxes are calculated, using the tax rate ϕ.

The discount factor i can be used as a profitability index, or IRR, when all investments and cash flows are discounted and NPV is set to zero. Knowing a desired profitability index, the IRR can be set to find the minimum, or levelized product price p which the product must be sold for. For the biofuel investor, this price might be indexed to the oil price, for example, to gauge investment risks associated with commodity price uncertainty.

Profitability of the private sector investment interpreted by the government, or to the public tax payer, is simply the present value of all tax revenues using a government discount rate of i less any investment grants or other government cash flows to the private investor, g:

$$NPV_{Gov.'t} = \sum_{j=1}^{N} (1+i)^{-j} \left(Q_j * p_j \right) \phi - g_j$$

(2)

Any action taken by the government which increases the present value of cash flows or reduces the present value of investments for the private sector affects the present value of tax income, or the economic value to the public.

4.4 Introduction to Paper III

Only one specific forest-biofuel product, Fischer-Tropsch diesel (FTD), is chosen for the economic case study for three reasons: i) it is "drop-in" ready with respect to infrastructure compatibility; ii) it has a low life cycle GHG emission profile; and iii) the long-term demand prospects for diesel substitutes in Norwegian road transport are significant. Technical risk is first quantified using statistical models of capital cost growth and plant underperformance (Merrow, et al., 1981). Impacts to public NPV and private IRR from an array of viable incentive policy instruments are then quantified. The goal of the study is to create an awareness of the range of outcomes that are associated with the policy instruments. Given specified performance criteria, policy cost-effectiveness with respect to private sector profitability and public policy goals (i.e, reduced GHG emissions and fossil-fuel dependency) is then assessed and quantified in light of a range of uncertain future oil prices.

4.5 Paper III

Incentivizing Wood-based Fischer-Tropsch Diesel Through Financial Policy Instruments: An Economic Assessment for Norway
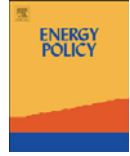
Incentivizing wood-based Fischer–Tropsch diesel through financial policy instruments: An economic assessment for Norway

Ryan M. Bright*, Anders H. Strømman

Industrial Ecology Programme, Department of Energy and Process Engineering, Norwegian University of Science and Technology (NTNU), 7491 Trondheim, Norway

ARTICLE INFO

ABSTRACT

The objective of this study is to evaluate a select set of financial incentive instruments that can be employed by the Norwegian government for encouraging early investment and production experience in wood-based Fischer–Tropsch diesel (FTD) technologies as a means to accelerate reductions in greenhouse gas emissions (GHG) stemming from road-based transport. We start by performing an economic analysis of FTD produced from Norwegian forest biomass at a pioneer commercial plant in Norway, followed with a cost growth analysis to estimate production costs after uncertainty in early plant performance and capital cost estimates are considered. Results after the cost growth analysis imply that the initial production cost estimates for a pioneer producer may be underestimated by up to 30%. Using the revised estimate we then assess, through scenarios, how various financial support mechanisms designed to encourage near-term investment would affect production costs over a range of uncertain future oil prices. For all policy scenarios considered, we evaluate trade-offs between the levels of public expenditure, or subsidy, and private investor profitability. When considering the net present value of the subsidy required to incentivize commercial investment during a future of low oil prices, we find that GHG mitigation via wood-FTD is likely to be considered cost-ineffective. However, should the government expect that mean oil prices in the coming two decades will hover between $97 and 127/bbl, all the incentive policies considered would likely spur investment at net present values $\leq$ $-100/tonne-fossil-$CO_2$-equivalent avoided.

© 2010 Elsevier Ltd. All rights reserved.

1. Introduction

Second generation biofuel produced from woody biomass is expected to be an effective avenue for reducing fossil fuel consumption and greenhouse gas (GHG) emissions in road transport (Bright and Strømman, 2009; Bright and Strømman, 2010; Bright et al., 2010; Edwards et al., 2007; van Vliet et al., 2009; Zah et al., 2007). The Norwegian boreal forest offers a large, underutilized source of woody biomass (Bolkesjø et al., 2006; Gjølsjø and Hobbelstad, 2009; Trømborg et al., 2008), and the Norwegian government is actively promoting the increased utilization of domestic forest resources for use as bioenergy (Trømborg and Leistad, 2009). While there are many application strategies which can efficaciously exploit the energy value of this resource – both within and outside the transport sector – the optimal strategy will vary depending on the primary policy objective(s) and/or sector(s) under target. This is demonstrated in (Joelsson and Gustavsson, 2010) and (Gustavsson et al., 2007) who show that oil use is more efficiently reduced in Sweden when

biomass replaces oil in stationary boilers rather than transport fuel produced in stand-alone plants, and similarly, that biomass usage outside the transportation sector may reduce GHG emissions more than biofuel in the transportation sector. It may also be the case that within the transport sector itself there are more effective uses of biomass resources for meeting GHG and energy reduction strategies. See for example (Ohlrogge et al., 2009; Campbell et al., 2009; Bright and Strømman, 2010). However, Grahn et al. (2009) show that industrialized nations cannot solely rely on reducing emissions from stationary sources and that biofuels become important for addressing options in transportation under scenarios involving stringent regionalized GHG emission caps, especially in the short- and medium-terms. This may be attributed to the difficulties in meeting near- and medium-term demands for rural road and heavy-duty freight transport in the absence of viable low-carbon alternatives.

In Norway, the government is aggressively targeting the road transport sector for the reduction of greenhouse gas emissions, and wood-based Fischer–Tropsch diesel (FTD) is viewed as an attractive part of the technological solution, particularly its use as a drop-in ready diesel substitute in rural and heavy-duty applications. Wood-FTD production technologies are soon scalable, with small-scale commercial production currently in the

---

* Corresponding author. Tel.: +47 73 59 89 72; fax: +47 73 59 89 43.
  E-mail address: ryan.m.bright@ntnu.no (R.M. Bright).

start-up phases (Kiener, 2008) and large-scale commercial production expected to commence as early as 2012 (IEA/OECD, 2008; Rudloff, 2008). Plans for a commercial operation producing 270 million liters/year of FTD by 2016 are on the drawing board (Green Car Congress, 2008; Xynergo, 2008).

Yet further progress in technological development and improvement in certain processing steps are still required in order to make FTD production more cost-effective (IEA/OECD, 2008; van Vliet et al., 2009; Zhang, 2010) and attractive to today's investors. In addition to high capital costs (IEA/OECD, 2008; Londo et al., 2010; van Vliet et al., 2009), barriers to short-term deployment include higher project risk because such technologies have yet to be proven at the commercial scale (IEA/OECD, 2008; Londo et al., 2010). However, a need to deploy advanced biofuel technology that can significantly contribute to reductions in fossil fuel use and GHG emissions, particularly those stemming from road-based transport, necessitates the execution of sound support policies designed to accelerate their early commercialization. To the extent that reductions in fossil fuel use and GHG emissions are intended to be achieved by means of alternative transport fuel, a clear focus needs to be placed on those alternative fuels, like wood-based FTD, that reduce global warming emissions (OECD, 2008). Only when new technologies like FTD are deployed can their volumes be scaled up, since one gains operational experience which leads to steadily decreasing production costs (de Wit et al., 2010). In the US, for example, corn ethanol production costs have decreased 62% since the earliest commercial-scale producers first entered the market around 1975 (Hettinga et al., 2009). Thus in order to steepen the learning curve in the short-term, early commercialization of FTD technologies will likely require, in addition to current environmental sustainability standards and quota mandates for biofuels in EU biofuel regulation (European Commission, 2009), economic support policies designed to remove market barriers and incentivize investment into specific technologies (OECD, 2008; Sandén and Azar, 2005). Incentive-oriented policy approaches whose purpose is generating technological change are likely to be important parts of the policy portfolio for addressing certain environmental problems like global warming (Jaffe et al., 2005).

1.1. Objectives

Given that the government has the goal of deploying specific technologies as a means to reach the two overarching policy goals of reduced fossil fuel dependency and GHGs emissions in road transportation, our primary objective in this study is therefore to evaluate a select set of financial incentive instruments that can be employed for encouraging investment in wood-FTD plants in the near-term. In this study, we do not concern ourselves with estimating cost reductions over time resulting from technological learning. Our goal is to quantify levels of economic support needed in the short-term in order to accelerate the deployment of commercial FTD technologies. This predicates an understanding of the production costs likely to be borne by pioneer producers which is inclusive of the inherent technological risks affiliated with 1st-of-a-kind, or pioneer commercial FTD plants. We start by performing an economic analysis of a pioneer commercial plant design, relying on capital and operating cost estimates reported in publically available literature. These estimates make use of optimized operating parameters, mass and energy balances, and conversion efficiencies exhibited at laboratory and/or pilot/demonstration scale – combined with scale-dependent installation factors – for estimating direct and indirect capital investment costs and annual operating and maintenance (O&M) expenditures of a large scale commercial FTD plant.

History shows us that production costs upon start-up are often higher than original estimates for pioneer commercial energy and chemical process plants integrating new technologies and processes due to unforeseen capital cost growth and under-performance (Merrow et al., 1981; Deutmeyer, 2010). Misestimation of the capital costs and performance of innovative energy process plants like wood-FTD plants can create problems for government and industry in planning the development and commercialization of pioneering plants. We therefore derive new production cost estimates by performing a cost growth analysis based on our initial cost estimates of the pioneer case so that local decision makers can make better planning and investment decisions in the short-term. We follow this with a sensitivity analysis in order to observe changes in the new FTD product cost that are associated with variances in select underlying financial assumptions and economic performance parameters. This is succeeded by an analysis of various financing schemes and economic support mechanisms that would be required to encourage private investment in the short-term. Finally, we then use the new cost estimate inclusive of the added-risks elements associated with FTD produced at a pioneer commercial plant in order to evaluate GHG abatement costs of seven government deployment policy scenarios in light of uncertain future oil prices.

2. Technology description

We choose a FTD process developed by CHOREN[TM] Industries for our analysis because the same process will resemble commercial FTD production in Norway in the short term (Green Car Congress, 2008; Xynergo, 2008), and, from a technical maturity standpoint, it is one of the most advanced BTL processes in the world (IEA/OECD, 2008). The process is based on 3-step gasification of woody biomass followed by Fischer–Tropsch synthesis into synthetic diesel. The process is appealing because it is highly versatile to varying feedstock compositions; however, this requires some novel technologies that today are unproven commercially. A more detailed description and review of CHOREN[TM]'s and other state-of-the-art FTD technologies can be found in Althapp et al. (2007), Blades et al. (2005), IEA/OECD (2008), van Vliet et al. (2009), Vogel et al. (2007), and Zhang (2010). The plant's processing steps can be aggregated into seven major block areas: *biomass treatment*, *gasification*, *gas cleaning*, *gas conditioning*, *Fischer–Tropsch synthesis*, *upgrading*, and *utilities*.

3. Methods and data

Commercial FTD production in Norway is based on CHOREN[TM] Industries' "Self-sufficient" process design operating on mixed forest residues. Material and energy balances for a commercial (500 MW$_{th\ input}$) FTD plant are based on design data for a 43 MW plant (β-plant) taking into account measured performance data for an existing 1 MW pilot plant (α-plant) operating in Freiberg, Germany, since 2003 (Althapp et al., 2007; Baitz et al., 2004). In our economic analysis, a discounted cash flow rate of return framework is employed to derive and compare a levelized FTD production cost at a pioneer plant in Norway both with and without cost growth analysis. A levelized production cost refers to the minimum price at which a unit of FTD must be sold for the project to break even, taking into account lifetime expenditures, revenues, capital investments, and return on investment. We henceforth refer to the "levelized" production cost as simply product cost; the pioneer case without cost growth analysis as our pioneer, starting point case ("Pioneer, SP"); and the case where

Table 1
Financial and performance assumptions for the starting point ("SP") and cost growth ("CGA") pioneer cases.

Parameter	Assumption
Analysis lifetime	20 years
Financing	100% equity
Construction period	Capital investment spread over three years: 8%, 60%, 32%
Inflation rate	1.2%
Post-tax IRR	10%
Fixed asset depreciation schedule	Double declining balance—utilities, 20 years; rest-of-plant, 7 years
Working capital	15% of annual change in operating and maintenance (O&M) expenditures
Salvage value	10%
Currency	2008 USD
Capacity factor	96%
Corporate tax rate	41%
Feedstock	Mixed forest residues, chipped, $60/tonne at 35 wt% moisture (16 GJ (LHV)/tonne) at plant gate
Conversion efficiencies	Biomass-to-FTD, 45% (LHV); overall, 54.3% (LHV)
Co-product prices	Naphtha, $0.50/l; electricity, $0.06/kWh

new estimates are derived after cost growth analysis as "Pioneer, CGA". Common financial and performance assumptions for both cases are presented in Table 1.

Fixed capital investment (FCI) together with O&M cost estimates for the starting point pioneer plant design are obtained from Vogel et al. (2007). FCI includes the sum of direct and indirect costs associated with a 500 MW$_{th}$ BTL plant based on UET/CHOREN™ Industries' Carbo-V® gasification and Shell's Middle Distillate Synthesis technologies currently employed at the beta plant. R&D costs, land costs, and decommissioning/demolition are not included. O&M costs are also obtained from (Vogel et al., 2007) and include fixed and variable operating costs. Non-feedstock variable operating costs include the cost of auxiliary materials (i.e., make-up water, bed materials) and residuals (i.e., fly ash/slag disposal, wastewater treatment). Fixed operating costs include costs for service and operation, personnel, insurance, administration, and "others" including fees or testing costs. FCI and O&M costs (2006€) are converted to constant 2008 US$ using an exchange rate of 0.8€/US$ (European Central Bank, 2009) and inflation rate of 1.2% (European Central Bank, 2009). Feedstock costs of $60/tonne (65% DM) are the "as-delivered" price and represent the average price for spruce-dominated forest residues suitable for use as bioenergy in Norway. Feedstock costs (Trømborg et al., 2008) along with the annual average wholesale prices for co-products naphtha (Statistics Norway, 2009a) and electricity (Statistics Norway, 2009b) supplied to the Norwegian market are indexed to 2008 USD using an exchange rate of 6.5 NOK/US$. A project contingency of 20% is included (Weyerhaeuser, 2000), raising total capital investment (TCI) costs to US $647.6 million (2008). In our discounted cash flow analysis for the pioneer, SP plant design, we assume a 6-month start-up period with revenues at 50%, variable operating and feedstock costs at 75%, and fixed operating costs at 100% of normal.

3.1. Cost growth analysis

Advanced process concepts tested only at the pilot- or demonstration-scales lack the large-scale commercial operating experience required to verify predictions of performance and cost (Frey et al., 1994). Because a FTD process of 500 MW$_{th\ input}$ scale has yet to be commercially deployed, the risks affiliated with performance and engineering design uncertainties may arise. We

perform a cost growth analysis accounting for these uncertainties for a pioneer commercial plant following a methodology developed by the RAND Corporation (Merrow et al., 1981). Beginning in 1979, a comprehensive study by RAND Corporation, under contract with the US Department of Energy (DOE), examined the accuracy and reliability of initial cost and performance estimates for innovative process plants generated for investment decision purposes. This effort was undertaken to understand and quantify the causes of cost growth in innovative chemical/energy process projects. The RAND study authors used cost engineering knowledge of the day and RAND client input (34 private sector clients providing 106 cost estimates) as a starting point for statistically isolating the factors that are strongly related to capital cost growth and performance shortfalls in pioneer process plants. The study collected planning and cost performance information on actual completed pioneer process plants in North America and used regression analysis techniques to find causal drivers of cost growth. Their analyses identified two sources of cost growth in energy and chemical process plants: a capital cost estimation that is too low, and a plant performance that is less than expected. This led to the development of two multi-factor linear regression models which facilitate the estimation of both capital cost overrun and plant underperformance. The models were highly accurate, as nearly one half of the estimates were predicted within plus or minus five percentage points of their actual cost growth. These models have been useful both to the US DOE and industry in making decisions about commercialization and about required subsidies and risks for synthetic fuels and other energy process plants.

We estimate cost growth by applying the two regression models to estimate the unexpected *capital cost growth* and *reduced plant performance* associated with the pioneer FTD plant, relying on detailed process and technology descriptions obtained from publically available literature for determining the values entered as the model's independent variables, described below. For more information about the model including detailed descriptions of their parameters, see (Merrow et al., 1981; Prasad, 2009). Similar approaches have been performed in the cost growth analysis of pioneering ethanol (Kazi et al., in press; Riley, 2002) and coal-to-liquids technologies (Bartis et al., 2008; Camm et al., 2008), as well as for gauging the uncertainty distribution of construction cost estimates for various new electricity generation technologies (Dowlatabadi and Toman, 1990).

3.1.1. Capital cost growth

The RAND study first conducted an extensive statistical analysis of *capital cost growth* for a sample of 40 process plants and 106 original cost estimates for developing a model of capital cost misestimation. They found that cost-underestimation is systematically related to lower levels of project definition and larger amounts of unproven technology. The problem is termed "cost growth" because the final capital costs incurred in designing and constructing the plants almost always exceeded the original cost estimates (Merrow et al., 1981), with over 70% of the total variance in misestimation accounted for by three parameters: (1) the percent of the cost estimate for equipment not demonstrated at the commercial scale; (2) the potential for impurity buildup; and (3) the level and quality of site-specific information that is included in the cost estimate. We assign a mid-range value for (1) as the specific technologies used in block areas *biomass treatment*, *gasification*, *gas cleaning*, and *gas conditioning* are – at the time of estimate – unproven at the commercial level (de Wit et al., 2010). Together, their share of capital costs relative to FCI is 45%. The variable accounting for potential buildup of impurities (2) was given a mid-range value. Excessive slag buildup in the

combustion chamber of the gasifier can affect operating pressure and temperature leading to reduced syngas production efficiencies, and further, the syngas may contain small traces of contaminants that could be harmful to downstream Fischer–Tropsch catalysts (Blades et al., 2005). On a scale of 0–5 – with a value of 5 correlating to a high probability of impurity buildup – we adopt a value of **2.5**. We give a medium value to the variable for project definition, as the level of site-specific and engineering information available at the time of estimation is assumed to be known.[1] Because it is unknown whether site-specific soils/hydrology together with health, safety, and environmental data were factored into the original estimate (see Merrow et al., 1981 for details on this factor), we give a value of **5** on a scale of 2–8, with 8 representing the lowest level of project definition.

When inserting the values into the capital cost growth regression model, we obtain a value for the estimated percentage of *capital cost growth* of **79%**. FCI of the pioneer SP plant case is divided by the cost growth percentage derived above to obtain a new FCI estimate for the pioneer CGA case. Additionally, to account for the greater uncertainty in equipment and other costs even prior to considering cost growth, project contingency is increased to 30% for the pioneer plant, raising TCI to US $871.6 million (2008). Sensitivity analysis of the independent variables contributing to capital cost growth can be found in the Appendix.

3.1.2. Reduced plant performance

Using data from 44 process plants, a second linear regression model developed by RAND Corporation (Merrow et al., 1981) is used to estimate production shortfalls of the pioneer CGA plant case. Production shortfalls are reductions in performance below desired operation capacity occurring after the initial start-up period, which RAND found strongly correlates with the introduction of new technologies. Parameters used to estimate reduced plant performance are based on: (1) the number of process steps that have not been demonstrated commercially; (2) whether the plant handles solids; (3) the percentage of actual mass and energy balance data validated with commercial-scale data; and (4) the severity of problems encountered in the development of waste handling. These four variables accounted for 90% of the observed variation in plant performance in the RAND study (Merrow et al., 1981). In our case, there are 4 process steps not demonstrated commercially: *biomass treatment*, *gasification*, *gas cleaning*, and *gas conditioning*, thus a value of **4** is given as the value for (1). Since the FTD plant handles solids, a value of **1** denoting "yes" is given for (2). Since, at the time the mass and energy balance data were compiled by (Althapp et al., 2007; Baitz et al., 2004) only 3 of the 7 process steps had been validated by commercial-scale data, a low value for (3) is given (**43%**). On a scale of 0–5, with 5 representing the highest severity level for unforeseen waste handling developments, we adopt a mid-range value of **2.4**, which is the mean value reported by 15 "solids plants" that participated in the RAND study (Merrow et al., 1981). Inserting our values into the second linear regression equation we obtain an estimated value for *reduced plant performance* of **21%**. Sensitivity analysis of the independent variables contributing to reduced plant performance can be found in the Appendix.

First-year FTD sales, variable operating costs (including feedstock costs), and co-product revenues of the pioneer SP plant are multiplied by the percentage of reduced plant performance to account for the reduced performance of the pioneer plant comprising our CGA case. For the discounted cash flow analysis, plant performance is increased by 25% per year until the desired capacity factor (96%) is reached.

[1] Using the beta-plant site in Freiberg, Germany, as a proxy.

3.2. Deployment policy analysis

Designing and evaluating prospective financial incentive packages that cost-effectively promote early production experience with wood-FTD in the face of significant uncertainty about the future represents a formidable challenge. There are numerous policy instruments available, and in all likelihood the best policy for a government to pursue will involve a combination of several unique instruments in the form of a package which is designed to ensure a relationship that yields both investor and government behavior that generates a set of cash flow exchanges over time (Camm et al., 2008). It is not within our objectives to review and describe the complete spectrum of individual policy instruments in detail, as comprehensive analyses of such instruments and how they can be applied to promote early commercialization of unconventional liquid fuels have been recently discussed in (Bartis et al., 2008; Camm et al., 2008; Sandor et al., 2008). However, Bartis et al. (2008) illustrate that for purposes of policy analysis, government incentives can be generally viewed as falling into one of five categories: (1) purchase guarantees; (2) price floors; (3) subsidies that reduce the private firm's investment cost; (4) subsidies that reduce the private firm's operating costs or increase revenues; and (5) government loan or loan guarantees for a portion of the firm's debt financing. In addition, income sharing agreements can be combined with any of the above incentive(s) when oil prices are high so that the government can be compensated for the associated costs and risks (Bartis et al., 2008; Camm et al., 2008).

We explore deployment scenarios that incorporate only those policy instruments (or package of instruments) capable of being assessed using our discounted cash flow framework, which include those resting within categories 2–5 (Table 2) as well as income sharing agreements. Our purpose is to illustrate their desirable attributes and not necessarily prescribe or recommend any one particular instrument or package, especially since we have no way of knowing what an investor might consider to be a desirable hurdle rate, nor do we have a way of knowing how

Table 2
Policy packages considered in our deployment policy scenario analysis.

Policy package name	Main instrument category	Description and model input assumptions
A	2	Price floor at $123/bbl; income sharing when oil hits $140/bbl via linearly increasing income/corporate tax hikes: 46% at $149/bbl, 51% at $159/bbl, 56% at $169/bbl, 61% at $180/bbl
B	3	10% capital investment grant+price floor
B2	3	50% capital investment grant+price floor+income sharing after oil price reaches $100/bbl, at which point corporate tax level is increased linearly at rates reaching 61% at $180/bbl
C	5	10-year government loan, 50% debt–equity ratio, 6% interest rate (assume no default)+price floor
C2	5	Loan guarantee (10-year term), 25% debt–equity ratio, 4% interest rate (assume no default)+price floor
D	4	Blender/distributor carbon-tax credit linked to actual displaced fossil carbon (current carbon tax on diesel=$0.08/l)+price floor
D2	4	Blender/distributor carbon-tax credit linked to actual displaced fossil carbon+equal non-carbon excise duty relief+price floor

much the government may "value" the new technology in metrics that cannot be expressed with economic indicators.

In our policy scenario analysis we consider only the pioneer CGA plant case and adopt the same set of financial assumptions used in our product cost assessment (Table 1) unless specified otherwise in Table 2. We use two performance metrics: real after-tax internal rate of return (IRR) for a private investor, and real net present value (NPV) of cash flows to and from the government. For measuring the cost-effectiveness to the government of alternative wood-FTD incentive packages, we calculate the cost to the government of increasing the IRR by one percentage point. To assess the value of this cost-effectiveness metric for any policy change, we introduce the policy change, measure how the change alters IRR values and government NPV, and divide the change in NPV by the change in private IRR. This allows us to compare the government's costs of increasing private IRR in different ways to specific incentive-package changes. When assessing cash flows to and from the Norwegian government, we use the Norwegian Finance Ministry's recommended discount rate of 6% for use in socio-economic analyses (Norwegian Finance Ministry, 1999). All scenarios incorporate a price floor set at an oil price equivalent which ensures a 10% real post-tax IRR to a private investor. Determining the price floor first involves assessing the policy effects without it. The change in NPV is then reassessed at oil prices below the floor.

When considering uncertainty, we limit our assessment to average oil price over the 20-year analysis period and exclude uncertainty regarding future feedstock and carbon-prices, as future oil price is likely to be the dominant uncertainty factor affecting an unconventional liquid fuel production investment decision (Bartis et al., 2008). The results from our scenario analyses are presented such that the effects on government NPV and private investors are shown in relation to a range of uncertain future oil prices. Except for the debt-financed – or "C" incentive packages – we assume all cases are 100% equity financed.

3.3. Costs of GHG abatement

Well-to-wheel global warming emissions associated with FTD in Norway are adopted from our previous work (Bright and Strømman, 2010; Bright et al., 2010) in addition to emissions for conventional diesel (Bright et al., 2010). Costs of GHG abatement attributed to any of the policy scenarios considered can be expressed as government NPV required to avoid fossil-diesel GHG emissions. When NPV is negative, this is in essence the level of public spending required to subsidize FTD deployment in order to obtain emission mitigation objectives affiliated with road transport. Expressing the cost-effectiveness of emissions reduction associated with any future FTD deployment policy can thus be expressed as

$$NPV, \text{1 tonne GHG avoided} = [(NPV_i)/(GHG_{Diesel}-GHG_{FTD})] \qquad (1)$$

where NPV_i is the NPV of the government policy yielding an investor IRR i at its corresponding average oil price over the 20-year analysis period, GHG_{FTD} the well-to-wheel GHGs of wood-FTD over the 20-year analysis period, and GHG_{Diesel} the well-to-wheel GHGs of the diesel reference fuel over the same 20-year period.

It is important to mention that all GHG abatement costs presented in subsequent sections of the article are inclusive of the current CO_2 tax on diesel in Norway, as this tax is currently applied to today's biodiesel and because it is unclear at this time whether or not more advanced biofuels like wood-FTD will be exempted in the future (Norwegian Finance Ministry, 2009). The effects of exempting FTD from this tax on private investor

profitability and government NPV are illustrated in policy scenario "D".

4. Results

4.1. Product cost: starting point and cost growth analysis pioneer cases

The result presented in Fig. 1 of our pioneer SP case indicates a product cost of $0.95/LDE (liter-diesel-equivalent). After correcting for exchange rates and inflation, we benchmark this result with five others reported in literature for near-term wood-based FTD (de Wit et al., 2010; Hamelinck and Faaij, 2006; Hamelinck et al., 2004; Tijmensen et al., 2002; van Vliet et al., 2009) and find it to be on the high side (+21–86%) for a plant of $\geq 400\ MW_{th}$ scale. This is not unexpected considering that the conversion processes and technologies are not identical and that results are contingent on a range of financial and economic assumptions (i.e., discount factor, capital and O&M costs, length of analysis period, feedstock costs, co-product prices, exchange rates, inflation rates, income tax rate, contingency factor, length of construction period, etc.)—as well as plant performance assumptions (i.e., start-up time, capacity factor, conversion efficiencies, main- and co-product yields, etc.).[2] Of the literature reviewed in our benchmarking analysis, only one study considered the CHOREN configuration which closely resembles the one of this study (van Vliet et al., 2009). Normalizing the cost from van Vliet et al. (2009) to ours, we find it to be 39% higher which can be expected due to the significant difference in scale (80 MW_{th}) resulting in lower economies of scale.

Compared to the SP plant design, product cost of the CGA case is 30% higher, shown in Fig. 1. Due to the increased capital investment expenditure and reduced plant performance, higher revenues are required to yield the same internal rate of return (IRR) over the life of the project, thus taxable income levels increase for the CGA case resulting in an increase in the amount of corporate/income tax that must be paid relative to the SP commercial plant. This increase comprises 31% of the total cost increase. The 10% higher project contingency factor notwithstanding, the estimated capital cost growth for the CGA case contributes $0.06/LDE in additional costs—or a 40% increase compared to the SP pioneer plant. Surprisingly, relative to TCI, the O&M costs (sum "Variable, Feedstock, and Fixed") are substantially larger, comprising around 50% and 60% of the total product cost for the CGA and SP plant, respectively. Operating and feedstock costs for the CGA case increase slightly as a result of the reduced plant performance occurring in the first three years after start-up.

FTD cost sensitivity to changes in input variables for key financial and economic performance parameters is assessed, with sensitivity represented by the green bars in the lower half of Fig. 1. A 20% change in TCI and IRR from the base case values used in our discounted cash flow model for the CGA case (values in y-axis labels, Fig. 1) results in an 11% and 12% change in product cost, respectively. A reduction in plant performance of three additional weeks per year downtime – or a lowering of the capacity factor to 85% from 96% (two weeks down time) – increases the product cost $0.14/LDE – or 10%. A $10/tonne

---

[2] It is strongly advised that the reader become familiarized with our inputs, assumptions, and modeling choices when interpreting our results. Refer to the tornado diagram of Fig. 1 to obtain a general understanding of the sensitivity effects our inputs and financial assumptions have on FTD product cost. Additionally, it is not clear whether or not a FTD-diesel price adjustment has been performed for normalizing heating values in the referenced studies included in our benchmark analysis.
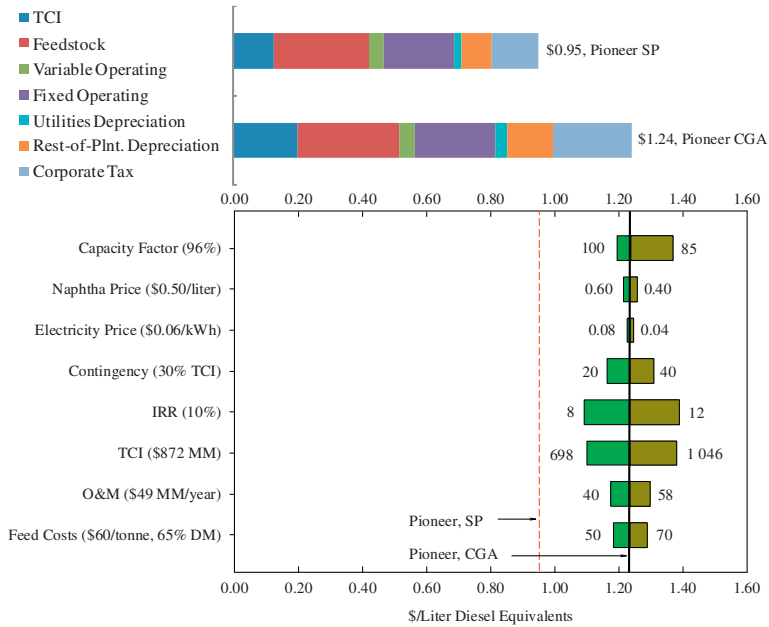
Fig. 1. FTD product costs in constant 2008 USD scaled to liter-diesel-equivalents (LDE) for the SP and CGA pioneer case, disaggregated by cost contribution parameter (top). Tornado diagram of FTD cost sensitivity to changes in input values (baseline values shown in y-axis, bottom figure).

increase (17%) in the feedstock price increases the product cost 5%, a moderate change. Changes in non-feedstock operating costs (O&M) and project contingency also results in moderate changes in the product cost. The FTD cost is least sensitive to changes in the price received for co-products naphtha and electricity, or with price changes in the enveloping energy market.

Fig. 2 illustrates the relationship between investor profitability (expressed as the anticipated IRR) and world oil price[3] for both the SP and CGA pioneer cases. To realize the 10% IRR assumption used in our product cost analysis, a difference of $29/bbl in mean oil price is needed for the pioneer CGA case over the SP case, suggesting that significant subsidies would be required to spur investment in a future of sustained low oil prices (assuming that an investor has a hurdle rate of 10%).

4.2. Policy scenario analysis

Fig. 3 covers the pioneer CGA case with no incentives in place across the same range of oil prices as in Fig. 2 (averaged across the 20-year analysis period) but shows the effect on both private investors (via real IRR) and government (via real public NPV), illustrating how variations only in oil prices characterize large uncertainty about the future (black curve).

As oil price climbs above $123/bbl, which is the minimum FTD-equivalent price needed to ensure the 10% IRR of our CGA case, both investor profitability and government NPV in the form of additional tax revenues would increase; and contrarily at lower oil prices, the investment opportunities appear less attractive to

the private investor and government NPV begins to reflect a public cost as it becomes negative. Knowing the lifetime GHG emission savings afforded by FTD (based on total production) and change in NPV at any given IRR and corresponding oil price, we are able to express public mitigation "costs" per tonne-GHG avoided as a function of the base case IRR in NPV terms (green curve, Fig. 3) following Eq. (1). In other words, if the government sets a minimum threshold IRR which it deems sufficient to spur private FTD investment – say 10% – the effectiveness of any future mitigation policy thus becomes a function of a changing IRR and the corresponding change in NPV required to mitigate 1-tonne-CO_2-equivalent at any oil price lower than that required for the investor to realize the threshold IRR. Thus the public mitigation cost of any subsidy policy can be expressed as negative NPV/tonne-CO_2-eq.-avoided at low oil prices (green curve, Fig. 3).

4.3. Scenario results

Guaranteeing an investor return of 10% (real after-tax IRR) in a scenario absent of any incentive mechanisms other than a price floor would require the price floor to be set at $123/bbl, shown in the top-left quadrant of Fig. 4 ("Policy Package A"). Should average oil prices hover around $72/bbl over the life of the project – the price floor subsidy would come at a significant government cost of around $570 million (−$570 MM NPV). However, if the government speculates that upon start-up we will return to a period of high oil prices (like the $147/bbl oil price experienced in the 3rd quarter of 2008), employing an instrument like an income sharing agreement to capitalize on such a scenario would be extremely effective as represented by the steep increase in NPV. The attractiveness of an income sharing mechanism is that the investor would still benefit, albeit via a slower rate of
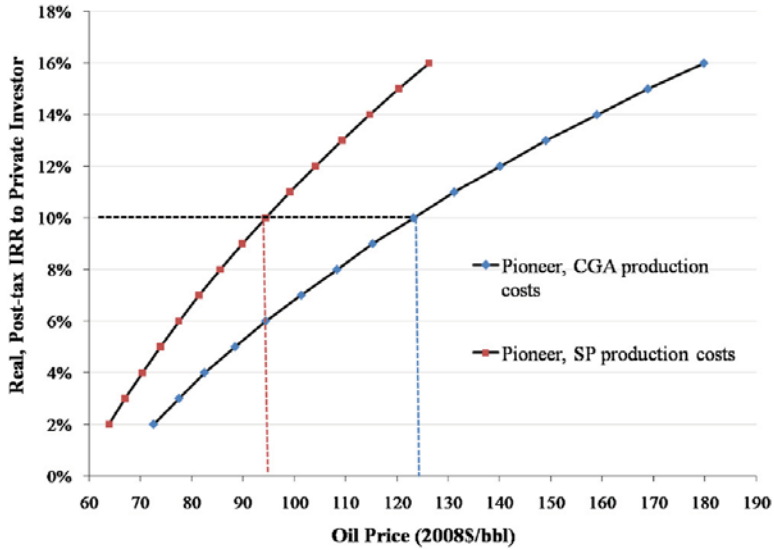
Fig. 2. Internal rate of return versus oil price, both pioneer cases.
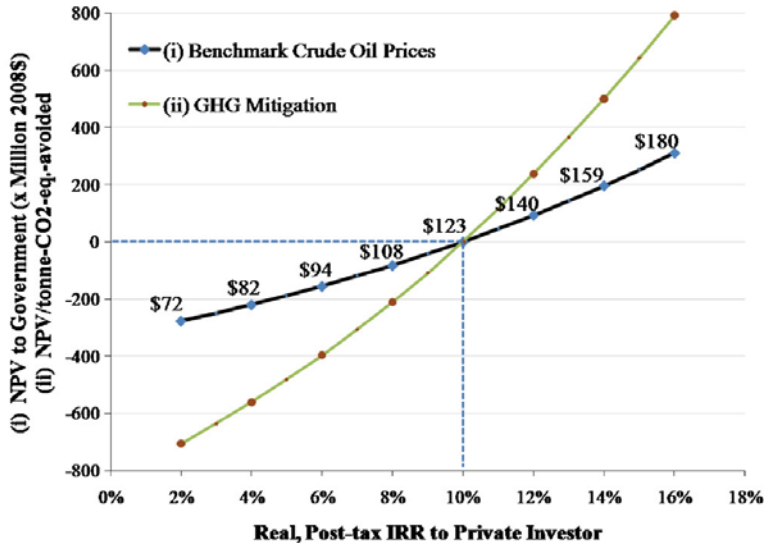
Fig. 3. Private and government effects of the pioneer CGA case with no policy incentives in place (black curve) and the cost-effectiveness of avoiding 1-tonne fossil GHG emissions (green curve). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

increasing IRR, should oil prices remain high. We illustrate in scenario "A" that the general effect of combining the two instruments increases the slope of the baseline curve which has the effect of escalating both downside risk and upside benefit for the government while having the opposite effect for investors.

Should the government employ an incentive package which retains the 10% IRR price floor in combination with a small capital investment grant, (package "B", bottom left, Fig. 4), the increase in

investor profits (and thus tax revenues) more than offsets the additional subsidy costs. This is reflected in the lowered price floor ($114 versus $123) which results in slightly higher NPV at lower oil prices relative to scenario "A." By increasing the capital investment grant to 50% of TCI (scenario "B2"), we see that this effect is even more pronounced: IRR is significantly enhanced to the point where the price floor can be set as low as $79/bbl. By incorporating an income sharing agreement after oil prices hit $100/bbl, the government can ensure that the cost of the subsidy
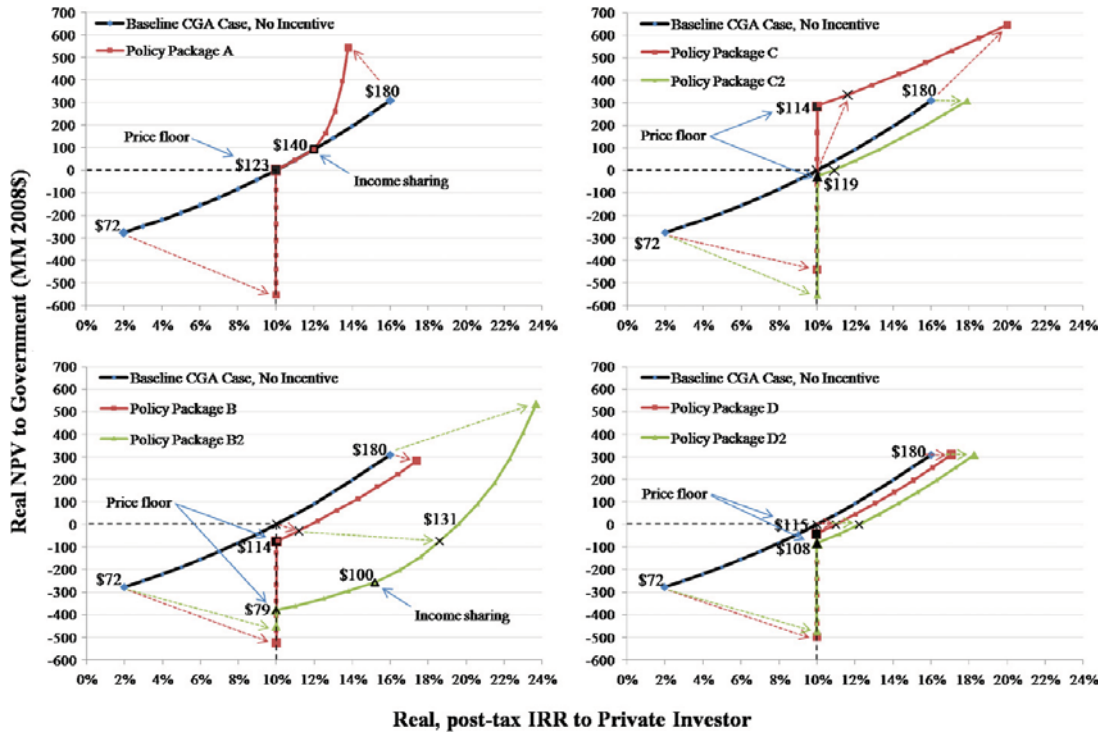
Fig. 4. Change in real IRR versus change in real NPV post-introduction of the seven deployment policy packages relative to the baseline pioneer CGA case (black curves). Dots along each curve correspond to the same oil prices as those shown in Fig. 3.

can be recovered at must faster rates and fully recovered at an oil price of $131/bbl. Our general finding here related to investment subsidies like capital investment grants is that benefits of increasing IRR come at relatively smaller costs (decreases to NPV) – or in other words, the base case curve tends to shift to the right along the x-axis more so than downwards along the y-axis.

Government-backed (loan guarantee) or direct-issued government loans are attractive in the sense that investors are able to leverage their capital through lower-interest debt-financing. This implies, however, that risk is ultimately transferred to the government. We consider both scenarios. Firstly, a loan administered directly by the government issued at a high debt–equity ratio and with an interest rate equal to the government discount rate (policy package "C" scenario), and secondly, a loan guarantee scenario where the government indemnifies a private lender which lends at a lower debt–equity ratio but at a lower interest rate ("C2"). Refer to (Camm et al., 2008) for an in-depth discussion on the drawbacks and benefits of government-issued and backed loans. As we illustrate in Fig. 4, in the event that an investor avoids default, a government issued loan has the effect of shifting the curve significantly upwards along the y-axis due to the net present value of interest paid on principle. As can be seen in Fig. 4, this scenario greatly reduces government risk as NPV turns negative only when oil prices are low due to the effects of the price floor, which is set at $114/bbl to ensure an investor IRR of 10%. With increases in oil price above the $114/bbl floor, however, IRR increases at a faster rate relative the base case, indicated by

the stretching of the red curve and the decreasing positive slopes of the red arrows (top right, Fig. 4).

Under the loan guarantee scenario of "C2", as indicated by the non-changing slope of the green arrow at oil prices above the price floor, IRR is increased with no adverse effect to the government. This is because the taxes previously paid on equity income are now paid by the lender whom we assume face the same tax rates as the private FTD investor. What we can infer from both scenarios is that debt financing allows the investor to increase its IRR as long as the cost of debt capital is below the IRR for the cash flows generated by the project. What we are not able to infer from our two debt-financed scenarios, however, is whether the resulting increase in IRR is more sensitive to the amount of debt financing (debt–equity ratio) or cost of debt capital (interest rate), although it would appear that the effect of increasing the former is greater than the effect of decreasing the latter.

In the event that a carbon tax credit is given to a downstream blender/distributor which is linked to the actual net reduction in carbon associated with substituting FTD for fossil diesel in Norway – such as in policy package "D" (red curve, bottom right quadrant, Fig. 4) – IRR increases with no adverse effect on government NPV when oil prices are above the price floor because the new tax revenue from FTD production offsets that foregone via the displaced carbon tax on diesel. Similarly, under scenario "D2" in which the duty relief is doubled for a downstream blender, IRR is increased even further, resulting in an even lower price floor. In essence, fuel tax relief has the effect of increasing investor revenues which shifts the entire curve in a positive

direction along the x-axis and lowers the price floor, which means the government would bear no risk at future oil prices above this floor.

4.4. Policy selection

Setting minimum financial performance criteria can assist in choosing the right policy. For example, if the government ruled out all policy instruments incapable of yielding an IRR of $\geq 10\%$ while mitigating fossil-GHG emissions at a public cost of $\leq \$100/\text{tonne-avoided}$ ($-\text{NPV}$), the range of incentive instruments or packages can be narrowed so that trade-offs between cost-effectiveness and investor attractiveness can be evaluated in light of uncertain future oil prices. We illustrate by plotting the policy cost-effectiveness of avoiding 1 tonne GHG emission in

order to isolate the threshold oil price at which the policy scenario meets the two performance criteria (Fig. 5).

We find that for all scenarios, none of the policy packages considered would qualify unless the government was confident that average oil price would meet or exceed \$97/bbl, which is the lowest threshold oil price found for any of the seven policy packages considered (package "C"). The other policy packages only begin to meet the government's cost-effectiveness criteria in a future where average oil prices span \$112–127/bbl, at which point IRR is extended over a range 10–19.2% for the other policy scenarios (at the corresponding threshold oil price).

Table 3 shows that for each dollar increase in oil price above the threshold oil price, IRR increases most under the two debt-financed scenarios (policy packages "C" and "C2"). Relative to policy package "C2", package "C" appears to be a more attractive
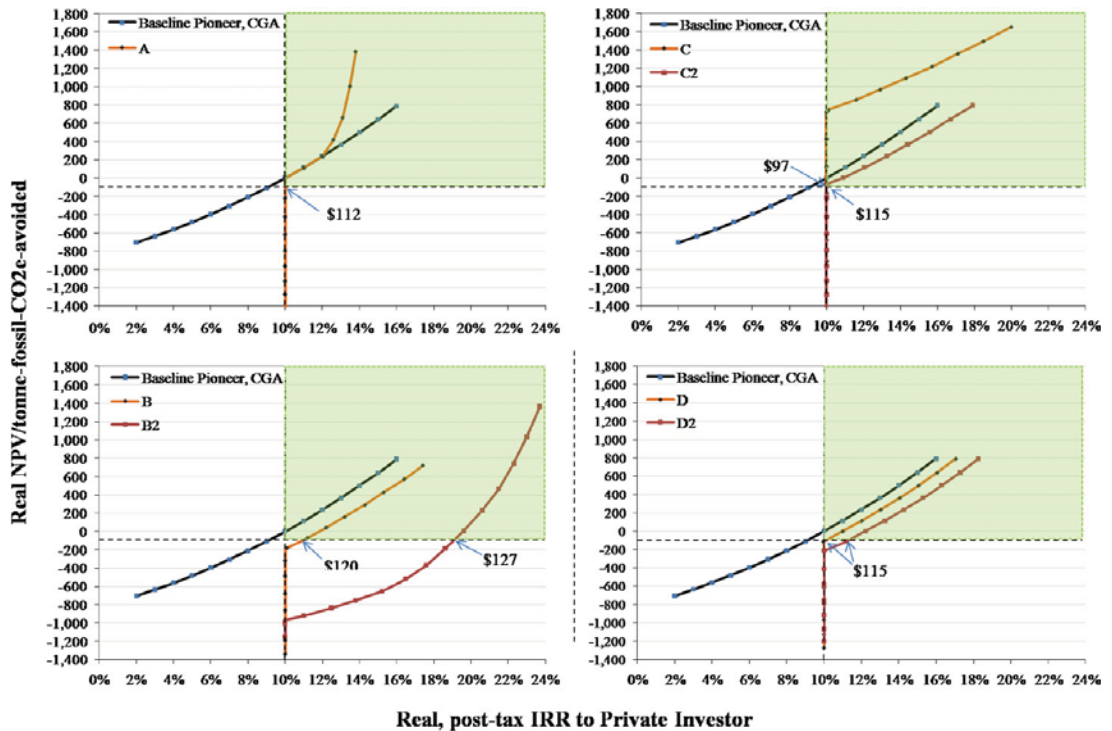
Fig. 5. Public mitigation costs (NPV) of GHG abatement versus private investor profitability (IRR). Green shading indicates the range of future mean oil prices which ensure that both the minimum NPV and IRR performance criteria of the prospective policy packages are met. The arrows point to the lowest possible mean oil price required under each scenario. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Table 3
Effects on policy cost-effectiveness (NPV) and investor IRR for each \$/bbl increase above or decrease below the threshold oil price.

	Threshold oil price (OP)	ΔIRR/\$ increase above OP	ΔNPV/\$ increase above OP	ΔIRR/\$ decrease below OP	ΔNPV/\$ decrease below OP
A	\$112	+0.06%	+\$22	−0.00%	−\$33
B	\$120	+0.10%	+\$14	−0.03%	−\$26
B2	\$127	+0.08%	+\$28	−0.17%	−\$19
C	\$97	+0.12%	+\$21	−0.00%	−\$41
C2	\$115	+0.12%	+\$14	−0.00%	−\$31
D	\$115	+0.11%	+\$14	−0.00%	−\$27
D2	\$115	+0.10%	+\$14	−0.03%	−\$27

policy for the government because NPV increases faster with each dollar increase above the threshold oil price of $97/bbl. However, should future oil prices dip below this value, this scenario also represents the riskiest alternative relative to the others in terms of the impacts on NPV. Should the government bet on a future of sustained low average oil prices that are lower than the minimum threshold prices shown in Table 3, none of the policy packages considered in our scenarios appear cost-effective, implying the need to re-design existing or consider new incentive packages altogether.

5. Discussion

We made use of a discounted cash flow framework in order estimate the product cost of wood-FTD associated with a pioneer commercial plant in Norway. We then assessed the potential for cost growth by applying two multi-factor linear regression models developed by the RAND Corporation (Merrow et al., 1981) for estimating capital cost growth and reduced plant performance occurring in the initial years after start-up, which led to a cost escalation of 30% and a final product cost of $1.24/LDE. We then evaluated the performance of a range of financial instruments that could be employed for incentivizing short-term private investment in order to accelerate reductions of fossil-based GHG emissions stemming from Norwegian road transportation. After introducing the policy package and by plotting the change in investor profitability (IRR) against public expenditure (NPV) relative to the cost growth pioneer reference case, we were able to identify those which served to enhance investor profitability while minimizing public mitigation costs at the lowest possible oil price.

Price floors were shown to boost investor confidence but presented high costs to government when oil prices were low. On the other hand, income sharing agreements compensated for the additional government risks when average oil prices were high. Blender tax credits linked to real carbon benefits had the effect of increasing investor profitability with no adverse effects on government NPV, as did government loan guarantees, but ensuring the 10% IRR resulted in the setting of a high oil price floor, below which the government began to incur additional costs. Investment subsidies such as capital investment grants were shown to increase investor IRR which allowed the setting of lower oil price floors, yet high oil prices were still required in order for the policy to be cost-effective. A government issued loan was found to be highly cost-effective at the lowest threshold oil price if one were to assume that the cost of administering the loan as well the risk of investor default were low. On the other hand,

however, NPV was found to decrease the most with each dollar decrease below the threshold oil price for this scenario relative to the other scenarios. Nevertheless, based on the results of our analysis, direct issued government loans were shown to be the most cost-effective financial incentive instrument considered in our analysis because both the minimum IRR and maximum allowable cost criteria were achieved at the lowest threshold oil price of $97/bbl.

It should be kept in mind that our analysis was not inclusive of the full spectrum of financial incentive instruments – like production subsidies, purchase guarantees, and accelerated tax depreciation schedules, for example – that should also be considered before a sound policy decision can be made. Additionally, other considerations regarding the public visibility of the subsidy, the costs of policy implementation, and the various risks elements concerning sunk costs or loan defaults need to be factored into future decision making. Further, while we have performed our assessment under the consideration of uncertain future oil prices and project costs, price uncertainties related to evolving energy markets and future carbon policies could also influence the cost-effectiveness of the deployment policy scenarios we have evaluated.

Nevertheless, we demonstrated that in addition to the technical uncertainties, perceived investor risk related to uncertain future oil prices poses a significant deployment barrier, and mitigating these risks when oil prices are low were shown to come at significant costs for the government. However, should the government expect higher average oil prices over the project life, a variety of financial incentive policies like those evaluated in this study can be implemented cost-effectively.

Appendix A

Fig. A1 shows the sensitivity of the input values used in the two multi-variate linear regression models for estimating capital cost growth and reduced plant performance. Greater sensitivity correlates with slope.

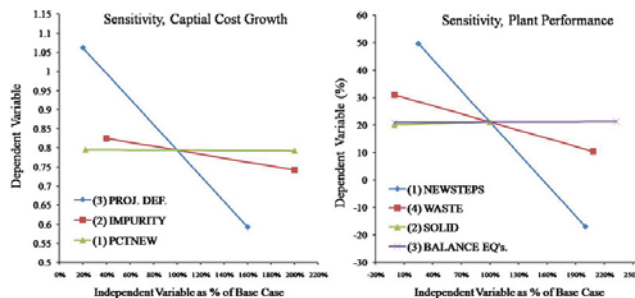**Fig. A1.** (Left) illustrates that cost growth is most sensitive to the independent variable chosen to represent the level of project definition ("(3) PROJ. DEF.") available at the time of estimate. (Right) illustrates that plant underperformance is most sensitive to the independent variable chosen to represent the number of new process steps not yet proven commercially ("(1) NEWSTEPS") at the time of estimate.

References

Althapp, A., Degen, W., Heinl, E., Rouveirolles, P., Rudloff, M., Stöhr, M., 2007. Del.: 1.23: SP1 Final Report, Version 4.3 (Final), Project: RENEW—Renewable Fuels for Advanced Powertrains. Umwelt- und Energietechnik (UET) GmbH, DaimlerChrysler AG, Volkswagen AG, Renault, B.A.U.M. Consult GmbH, Freiberg, Germany.

Baitz, M., Binder, M., Degen, W., Deimling, S., Krinke, S., Rudloff, M., 2004. Comparative life-cycle assessment for SunDiesel (CHOREN Process) and conventional diesel fuel: executive summary. PE Europe GmbH—Life Cycle Engineering.

Bartis, J.T., Camm, F., Ortiz, D.S., 2008. Producing Liquid Fuels from Coal: Prospects and Policy IssuesRAND Corporation, Santa Monica, CA.

Blades, T., Rudloff, M., Schulze, O., 2005. Sustainable SunFuel from CHOREN's Carbo-V process. ISAF XV. Accessed online September 18, 2009 at: ⟨http://www.choren.com/dl.php?file=San_Diego_-_Final.pdf⟩. San Diego.

Bolkesjø, T.F., Trømborg, E., Solberg, B., 2006. Bioenergy from the forest sector: economic potential and interactions with timber and forest products markets in Norway. Scandinavian Journal of Forest Research 21, 175–185.

Bright, R.M., Strømman, A.H., 2009. Life cycle assessment of second generation bio-ethanol produced from scandinavian boreal forest resources: a regional analysis for middle Norway. Journal of Industrial Ecology 13 (4), 514–531.

Bright, R.M., Strømman, A.H., 2010. Fuel-mix, vehicle technologies, and transport demand affect prospects for biofuels in northern Europe. Environmental Science & Technology 44 (7), 2261–2269.

Bright, R.M., Strømman, A.H., Hawkins, T.R., 2010. Environmental assessment of wood-based biofuel production and consumption scenarios in Norway. Journal of Industrial Ecology 14 (3), 422–439.

Camm, F., Bartis, J.T., Bushman, C.J., 2008. Federal Financial Incentives to Induce Early Experience Producing Unconventional Liquid Fuels. RAND Corporation, Santa Monica, CA.

Campbell, J.E., Lobell, D.B., Field, C.B., 2009. Greater transportation energy and GHG offsets from bioelectricity than ethanol. Science 324 (5930), 1055–1057.

Deutmeyer, M.W., 2010. Securing reliable long term solid biomass supply for large scale users. Presentation at 18th European Biomass Conference and Exhibition, May 6, 2010, Lyon, France.

de Wit, M., Junginger, M., Lensink, S., Londo, M., Faaij, A., 2010. Competition between biofuels: modeling technological learning and cost reductions over time. Biomass and Bioenergy 34, 203–217.

Dowlatabadi, H., Toman, M., 1990. Technology choice in electricity generation under different market conditions. Resources and Energy 12, 231–251.

Edwards, R., Griesemann, J.-C., Larivé, J.-F., Mahieu, V., 2007. Well-to-Wheels Analysis of Future Automotive Fuels and Powertrains in the European Context, Version 2c. EUCAR, CONCAWE, JRC.

EIA, 2010. US Energy Information Administration (EIA) Independent Statistics and Analysis: International Petroleum (Oil) Prices and Crude Oil Import Costs. Accessed online April 3, 2010 at: ⟨http://www.eia.doe.gov/emeu/interna tional/oilprice.html⟩.

European Central Bank, 2009. Statistical Data Warehouse. Accessed online November 4, 2009 at: ⟨http://sdw.ecb.int/home.do?chart=t1.9⟩.

European Commission, 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 and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC, Brussels.

Frey, H.C., Rubin, E.S., Diwekar, U.M., 1994. Modeling uncertainties in advanced technologies: application to a coal gasification system with hot-gas cleanup. Energy 19, 449–463.

Gjølsjø, S., Hobbelstad, K., 2009. Energipotensialet fra skogen i norge [The energy potential from Norwegian forests], Oppdragsrapport fra Skog og landskap 09/2009. Norsk institutt for skog og landskap, Ås.

Grahn, M., Azar, C., Lindgren, K., 2009. The role of biofuels for transportation in CO₂ emission reduction scenarios with global versus regional carbon caps. Biomass and Bioenergy 33 (3), 360–371.

Green Car Congress, 2008. Norske Skog/Xynergo and CHOREN to Cooperate on Biomass-to-Liquids in Norway. Accessed August 7, 2009 at: ⟨http://mixxed green.com/Biofuel/⟩.

Gustavsson, L., Holmberg, J., Dornburg, V., Sathre, R., Eggers, T., Mahapatra, K., Marland, G., 2007. Using biomass for climate change mitigation and oil use reduction. Energy Policy 35, 5671–5691.

Hamelinck, C.N., Faaij, A.P.C., 2006. Outlook for advanced biofuels. Energy Policy 34, 3268–3283.

Hamelinck, C.N., Faaij, A.P.C., den Uil, H., Boerrigter, H., 2004. Production of FT transportation fuels from biomass; technical options, process analysis and optimisation, and development potential. Energy 29, 1743–1771.

Hettinga, W.G., Junginger, H.M., Dekker, S.C., Hoogwijk, M., McAloon, A.J., Hicks, K.B., 2009. Understanding the reductions in US corn ethanol production costs: an experience curve approach. Energy Policy 37, 190–203.

IEA/OECD, 2008. From 1st- to 2nd-Generation Biofuel Technologies: An Overview of Current Industry and RD&D Activities. IEA Bioenergy. IEA, OECD, Paris.

Jaffe, A.B., Newell, R.G., Stavins, R.N., 2005. A tale of two market failures: technology and environmental policy. Ecological Economics 54, 164–174.

Joelsson, J.M., Gustavsson, L., 2010. Reduction of CO₂ emission and oil dependency with biomass-based polygeneration. Biomass and Bioenergy 34 (7), 967–984.

Kazi, F.K., Fortman, J.A., Anex, R.P., Hsu, D.D., Aden, A., Dutta, A., Kothandaraman, G., 2010. Techno-economic comparison of process technologies for biochem-ical ethanol production from corn stover. Fuel, in press, doi:10.1016/j.fuel.2010.01.001.

Kiener, C., 2008. Start-up of the first commercial BTL production facility—the beta-plant, Freiberg. Presented at16th European Biomass Converence & Exhibition Valencia.

Londo, M., Lensink, S., Wakker, A., Fischer, G., Prieler, S., van Velthuizen, H., de Wit, M., Faaij, A., Junginger, M., Berndes, G., Hansson, J., Egeskog, A., Duer, H., Lundbaek, J., Wisniewski, G., Kupczyk, A., Könighofer, K., 2010. The REFUEL EU road map for biofuels in transport: application of the project's tools to some short-term policy issues. Biomass and Bioenergy 34, 244–250.

Merrow, W.E., Phillips, K.E., Myers, C.W., 1981. Understanding Cost Growth and Performance Shortfalls in Pioneer Process PlantsThe RAND Corporation, USA.

Norwegian Finance Ministry, 1999. Circular R-14/99: Treatment of the discount rate, risk, and income tax expense in socio-economic analyses. Accessed online February 8, 2010 at: ⟨www.finans.dep.no⟩.

Norwegian Finance Ministry, 2009. Personal communication with Lovise Bauger, November 12.

Ohlrogge, J., Allen, D., Berguson, B., DellaPenna, D., Shachar-Hill, Y., Stymne, S., 2009. Driving on biomass. Science 324 (5930), 1019–1020.

OECD, 2008. Biofuel Support Policies: An Economic Assessment. Organization for Economic Cooperation and Development, Paris.

Prasad, R., 2009. AACE International Recommended Practice No. 43R-08—risk analysis and contingency determination using parametric estimating: example models as applied for the process industries. The Association for the Advancement of Cost Engineering (AACE), p. 12.

Riley, C., 2002. Bioethanol: a renewable transportation fuel from biomass, presenta-tion at AlChE spring conference. US Department of Energy, National Renewable Energy Laboratory, Accessed online November 13, 2009 at: ⟨http://www1.eere.energy.gov/biomass/pdfs/renewable_fuel.pdf⟩ Golden, Colorado, USA.

Rudloff, M., 2008. The CHOREN BTL process: status, progress, and future prospects. The RENEW Project—Renewable Fuels for Advanced Powertrains, Integrated Project for Sustainable Energy Systems, Brussels.

Sandén, B.A., Azar, C., 2005. Near-term technology policies for long-term climate targets—economy wide versus technology specific approaches. Energy Policy 33, 1557–1576.

Sandor, D., Wallace, R., Peterson, S., 2008. Understanding the Growth of the Cellulosic Ethanol Industry. NREL; The Peterson Group, Golden, CO, USA.

Statistics Norway, 2009a. Energy Statistics. Accessed November 2, 2009 at: ⟨http://www.ssb.no/english/subjects/01/03/10/energi_en/⟩.

Statistics Norway, 2009b. Manufacturer and wholesale price indices. Accessed November 2, 2009 at: ⟨http://www.ssb.no/vppi_en/⟩.

Tijmensen, M.J.A., Faaij, A.P.C., Hamelinck, C.N., van Hardeveld, M.R.M., 2002. Exploration of the possibilities for production of Fischer Tropsch liquids and power via biomass gasification. Biomass and Bioenergy 23, 129–152.

Trømborg, E., Bolkesjø, T.F., Solberg, B., 2008. Biomass market and trade in Norway: status and future prospects. Biomass and Bioenergy 32, 660–671.

Trømborg, E., Leistad, Ø., 2009. IEA Bioenergy Task 40—Country Report 2009 for Norway. Enova; IEA Bioenergy; Norwegian University of Life Sciences, Ås, Norway.

van Vliet, O.P.R., Faaij, A.P.C., Turkenburg, W.C., 2009. Fischer–Tropsch diesel production in a well-to-wheel perspective: a carbon, energy flow, and cost analysis. Energy Conversion and Management 2009, 855–876.

Vogel, A., Brauer, S., Müller-Langer, F., Thrän, D., 2007. Conversion Costs Calculation: Deliverable D 5.3.7, Project: RENEW—Renewable Fuels for Advanced Powertrains, Project: RENEW—Renewable Fuels for Advanced Powertrains. Institute for Energy and Environment (IE), Leipzig.

Weyerhaeuser, 2000. Biomass Gasification Combined Cycle, Final Report, DOE/GO/10173, Agenda 2020. Weyerhaeuser Company, Prepared for the US Depart-ment of Energy (DOE), Federal Way, Washington, p. 83.

Xynergo, 2008. Xynergo and CHOREN to Cooperate on Second Generation Biofuel. Accessed January 18, 2009 at: ⟨http://www.xynergo.com/press_releases/view/5⟩.

Zah, R., Heinz, B., Gauch, M., Hischier, R., Lehmann, M., Wäger, P., 2007. Ökobilanz von Energieprodukten: Ökologische Bewertung von Biotreibstoffen [Life Cycle Assessment of Energy Products: Environmental assessment of biofuels]. Empa, Bern, Switzerland.

Zhang, W., 2010. Automotive fuels from biomass via gasification. Fuel Processing Technology, 91(8), 866–876.

4.6 Uncertainties and Limitations

Uncertainty of policy outcome with respect to oil price is robust; however, fluctuations in the price of raw material inputs like forest biomass, the price of carbon, and in the price of co-products like electricity and naphtha will affect the product cost and thus the cost of the support policy. An additional uncertainty lay in the choice of financial and performance parameter assumptions. While parameter uncertainty was not included in policy analysis, the magnitude of such uncertainty may be gauged based on the results of the performed sensitivity analysis.

4.7 Summary and Conclusions

The study identified and quantified the private and public sector risks associated with one specific type of large-scale pioneer biofuel production technologies. The technical risk enveloping capital-intensive investments in novel energy process technologies like large scale forest-FTD correlated to a 30% increase in production cost assuming an IRR of 10%. This translated to an equivalent oil price of $123/bbl that must be sustained over a 20-year period beginning at start-up. At the current oil price (Brent index), this profitability standard is not met, thus financial incentives from the public sector would be required. Various policy instruments and combinations of instruments were shown to be effective at reducing private sector costs such that high profitability standards could be realized with minimal costs to the public sector (i.e., NPV from lost tax revenues, capital investment grants, etc.); however, only one of the policies considered in Paper III would be cost-effective at 2011 (~$90-100/bbl) oil prices given both performance standards of $\leq$\$100/tonne-GHG-avoided and $\geq$10% IRR to private investor.

4.7.1 Research Implications

If governments of Nordic Europe choose to play a more proactive role in facilitating the development of new industries by sharing the risk of novel technologies, additional investigation into other advanced forest-biofuel production technologies should be explored using similar analytical frameworks, and other economic risk elements not considered in Paper III (i.e., costs of administering loans, sunk costs, other bureaucratic costs, etc.) should be considered.

Chapter 5: Silver Bullet or Buckshot? Understanding Forest Biofuel's Niche in Regional Climate Friendly Road Transportation

5.1 Questions

In Chapter 3, avoided global GHG emissions due to fossil fuel substitution in Norway were quantified. While the domestic surplus biomass resource base was found to be large, growth in projected future demand for liquid fuels combined with the inefficient conversion of the energy in biomass to biofuel resulted in a limited fossil fuel displacement potential. It is clear that progress towards more climate-friendly road transport will require measures beyond single technical fixes like biofuels, which address only the emission intensity dimension. The other two factors contributing to transport-related GHG emissions – fuel intensity and overall transport activity – also warrant attention. Additionally, trade in forest products between Norway, Sweden, and Finland is significant, and since forest-biofuel production technologies and the structure of forest products industries of Sweden and Finland may differ from Norway, these regions ought to be included in the assessment. As a result, additional research questions remain unanswered and are the subject of this chapter:

- Given a finite resource potential throughout the entire Nordic Europe region, how might forest-biofuels fulfill shared policy objectives of energy independency and climate change mitigation when trends in current consumption patterns and vehicle technologies are extrapolated into the future?

- How might complimentary measures aimed at minimizing demand growth and reducing energy use per unit of transport service contribute to regional policy objectives?

- How effective might these additional measures be relative to policies focusing solely on fossil fuel substitution?

- What are the global climate change implications due to regional road transport consumption when the full extent of global supply chains are considered?

5.2 The IPAT Analogy

In the late 1960s, Paul Ehrlich and John Holdren illustrated a very simple identity known as "IPAT" governing the relationship between relative and absolute decoupling of growth and impact (Ehrlich & Holdren, 1969). When decomposing environmental impact (I) into its constituent factors – population (P), affluence (A), and technology (T) – it is clear that the technology factor must not be seen in isolation. For "absolute" decoupling of environmental impact and economic growth to occur, the rate at which the technology factor goes down must exceed the combined rate at which the other two factors increase. This identity serves as a useful analogy for better understanding how technologies like forest biofuel will contribute to regional policy objectives in the face of sustained economic and population growth.
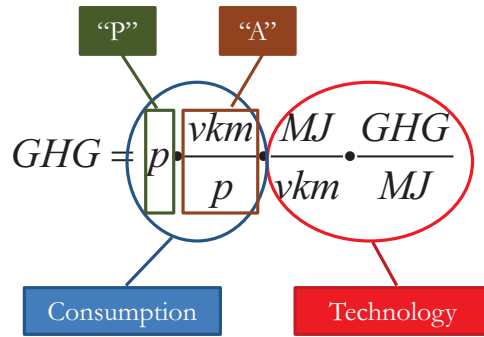
Figure 5. Road transport GHG emission as influenced by technology- and consumption-related factors. Consumption is decomposed for linking with the IPAT analogy. *"p" = passenger; "MJ" = megajoule; "vkm" = vehicle kilometer.* Note that *"p"* may be substituted with *"t"* when dealing with freight transportation, where *"t" = tonnes.*

Consumption of passenger and freight transport is influenced by the total number of passengers or goods needing to be transported by vehicles over a certain distance. One might associate the number of vehicle kilometers that are generated per passenger (or tonne) – "vkm/p" – as an indicator of affluence (A = \$/person), with total demand for transport activity (in vkm) as the product of the absolute number of passengers (or tonnage) and level of affluence. This is expressed as the first two terms comprising "Consumption" on the right side of the equation in Figure 5. Structural drivers affecting affluence (vkm/p) might be shaped by the degree of urbanization or rural development which might affect distances traveled or vehicle occupancy. Economic drivers affecting affluence are of course related to wealth (personal income) and the degree of discretionary income available for transport spending.

The technology variables shown in Figure 5 are related to vehicle fuel efficiency (MJ/vkm) and the GHG intensity of the fuel supply (GHG/MJ). Fuel efficiency is predominantly affected by engine size, vehicle weight, and aerodynamics, although it can be influenced by individual driving behavior. GHG-intensity – often synonymous with direct emissions per unit fuel combustion – ought to include indirect (life cycle emissions) from fuel production.

5.3 Applicability of MRIOA

As discussed in Chapter 3, hybrid IO-LCA frameworks make good frameworks for a controlled and transparent evaluation of technology change. The "hybrid" framework enables one to link explicit cause-effect relationships of specific technologies with environmental impact within the full scope of production and consumption activity occurring in a defined region. When the question requires a broad geographic scope to be taken, it is important that trade is accurately and fully represented, particularly for Norway, Sweden, and Finland – regions with significant trade both amongst themselves and with regions outside Nordic Europe. As mentioned in Chapter 3, for regions that are fairly open with respect to trade, the risk of not being able to identify consumption-driven emissions associated with foreign production is prevalent when imports are modeled using the domestic region's own technology as proxy. Multi-region input-output analysis (MRIOA) overcomes this problem and becomes preferable when dealing with multiple regions and with significant volumes of production and trade. A more accurate representation of

the climate impacts of consumption requires the full extent of global supply chains to be considered.

Full multi-region models endogenously combine domestic technical coefficient matrices with import matrices from multiple countries or regions into a single large coefficient matrix, thus capturing trade supply chains between all trading partners inclusive of feedback effects. "Feedback effects" are changes in production in one region that result from changes in intermediate demand in another region, which are in turn brought about by demand changes in the first region (Miller, 1969; Wiedmann, 2009). A distinction may be made between uni-directional MRIOA, where multiple trading partners are represented but the analysis is limited to trade flows in a single direction, and multi-directional trade MRIOA, where trade in all directions is considered (Lenzen, et al., 2004; G. P. Peters & Hertwich, 2009). The latter implies the inclusion of feedback loops and captures the direct, indirect, and induced effects of trade (G. P. Peters & Hertwich, 2009). Since the objective is to address consumption of the Nordic Europe region in its entirety, an analysis of multi-directional trade is required. In Chapter 3, since the focus was on Norwegian consumption alone, analysis of uni-directional trade from mainland Europe was sufficient.

5.3.1 Road Transport Scenarios and IPAT

In Chapter 3, only the GHG intensity parameter is linked to the policy scenarios. Answering the primary research questions formulated at the beginning of this Chapter require linking the "Consumption" parameters to the modeling framework in addition to the energy efficiency (MJ/vkm) "Technology" parameter. Since population is not a parameter in IO-based modeling frameworks, this is overcome by expressing "consumption" in units of pkm (or tkm for freight), an exogenously defined final demand variable that is adjusted in scenario analyses. For example, 10 passengers riding a bus for 1 km would equate to 10 pkm, or 1/10 vkm per passenger. If instead only 5 passengers choose to take the bus and the other 5 opt to take a single private car, we would still have 10 pkm but now we have a total of 2 vkm because two vehicles are needed, requiring more energy consumption which induces more emissions. The following equation illustrates the relationships between transport activity (vkm), occupancy rate (p/v), and vehicle ownership (v/p), which, when used together, indicate the total level of road transport consumption in any given region expressed as pkm:

$$pkm = \frac{p}{v} \bullet vkm \Leftrightarrow vkm = \frac{v}{p} \bullet pkm$$

(3)

Information on occupancy rates such as number of passengers per vehicle (p/v) are known and are thus used as an "Affluence" indicator. As previously mentioned, it is important to bear in mind that "Affluence" may be driven just as much by structural drivers as by wealth, although one could argue that wealth does have an influence in shaping transportation structure. Nevertheless, the term is adopted to be consistent with the IPAT analogy, and it is now possible to link affluence with population, and similarly, it now becomes possible to link energy intensity with energy efficiency associated with vehicle "Technology", illustrated in Figure 6.
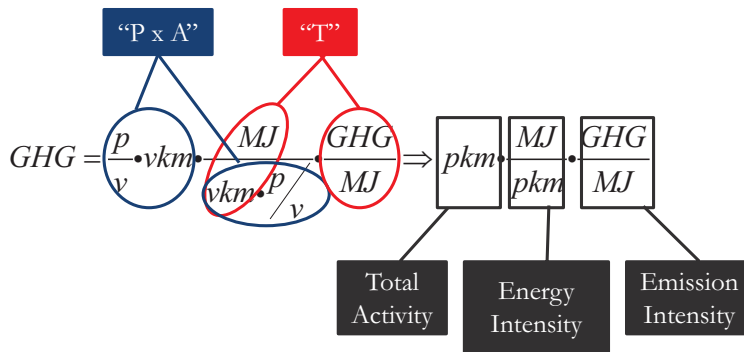
Figure 6. Total road transport activity is a product of population and affluence which drive consumption, or final demand. Affluence also plays a role in determining the intensity of energy use per unit transport service consumed, also influenced by vehicle technology (energy efficiency). Forest biofuels play a role in influencing the emission intensity of energy consumed in road transport. Total activity, energy intensity, and emission intensity are the three factors contributing to GHG emission from road transportation. The picture is the same for freight transport (substitute "p" with "t").

5.4 Introduction to Paper IV

The research questions require an analysis of the effects of policies which address road transport consumption (pkm, tkm) and energy intensity (MJ/pkm, MJ/tkm) in isolation from biofuel substitution effects (GHG/MJ). Regarding biofuel substitution, the regional resource base is approximated over the long-term so that an upper limit on biofuel production can be used to assess the degree of regional self-sufficiency given a policy scenario of maximum regional biofuel production. Parameter values for the reference scenario are derived via extrapolation of historical trends reported in statistics and from outputs of dynamic modeling studies of energy and transport in the Nordic Europe region.

Fuel-mix, Fuel Efficiency, and Transport Demand Affect Prospects for Biofuels in Northern Europe

Fuel-Mix, Fuel Efficiency, and Transport Demand Affect Prospects for Biofuels in Northern Europe

RYAN M. BRIGHT* AND
ANDERS HAMMER STRØMMAN

Industrial Ecology Programme, Department of Energy and Process Engineering, Norwegian University of Science and Technology (NTNU), 7491 Trondheim, Norway

Rising greenhouse gas (GHG) emissions in the road transport sector represents a difficult mitigation challenge due to a multitude of intricate factors, namely the dependency on liquid energy carriers and infrastructure lock-in. For this reason, low-carbon renewable energy carriers, particularly second generation biofuels, are often seen as a prominent candidate for realizing reduced emissions and lowered oil dependency over the medium- and long-term horizons. However, the overarching question is whether advanced biofuels can be an environmentally effective mitigation strategy in the face of increasing consumption and resource constraints. Here we develop both biofuel production and road transport consumption scenarios for northern Europe—a region with a vast surplus of forest bioenergy resources—to assess the potential role that forest-based biofuels may play over the medium- and long-term time horizons using an environmentally extended, multiregion input−output model. Through scenarios, we explore how evolving vehicle technologies and consumption patterns will affect the mitigation opportunities afforded by any future supply of forest biofuels. We find that in a scenario involving ambitious biofuel targets, the size of the GHG mitigation wedge attributed to the market supply of biofuels is severely reduced under business-as-usual growth in consumption in the road transport sector. Our results indicate that climate policies targeting the road transport sector which give high emphases to reducing demand (volume), accelerating the deployment of more fuel-efficient vehicles, and promoting altered consumption patterns (structure) can be significantly more effective than those with single emphasis on expanded biofuel supply.

Introduction

From 1990 to 2006, GHG emissions from the transport sector in the EU-15 grew 26%, representing 21% of total EU-15 GHG emissions, with more than 90% originating from road transport (*1*). This is indicative of the formidable challenges facing policy makers seeking to curb GHG emissions and transition to a less fossil-dependent road transport sector. Among the policy options considered are targets for replacing conventional fossil fuels with alternative energy carriers like biofuels, evidenced by the most recent legislative resolution of the European Parliament on the promotion and use of energy from renewable sources (*2*). More advanced biofuels,

particularly those made from lignocellulosic feedstocks such as agricultural and forest residues among others, typically offer greater GHG, health, and land-use benefits than their first generation predecessors (*3–7*)—including those produced from local forest-based feedstocks in Scandinavia (*8, 9*). In the Fenno-Scandinavian region (Finland, Norway, Sweden), the boreal forest provides a significant source of underutilized bioenergy (*10*), and the production of forest-based biofuels is an attractive policy avenue being explored. Yet the prevailing question is whether a stand alone biofuel policy is sufficient by itself in the progression toward more sustainable road transport. While energy intensities in both freight and passenger transport decreased at an average annual rate of 0.1% and 0.7% since 1990 (*11, 12*), respectively, average annual growth in total volumes of both passenger and goods transport throughout the region grew 1.8% and 3.1%, respectively (*13*). Additionally, growth in the volume of demand for private transport has outpaced that for public transport (*11–14*). Further, structural performance indicators such as average number of persons per private vehicle, taxi, and bus for passenger transport have been steadily falling and/or have remained unchanged; while for goods transport, the average number of vehicle kilometers per volume of goods requiring transport has been on the rise (*15–18*). Since a need for mobility is such an integral component of economic growth (*19*), and since effective consumption-based policy solutions are likely to meet with some resistance (*20–22*), the road transport sector continues to remain an "elephant in the room" in terms of addressing sectoral-based climate solutions, hence biofuel-oriented strategies have historically been viewed as more politically palatable, and as such, have received much of the policy attention to date.

Framework. Envisioning alternative futures, exploring plausible development pathways, and identifying factors conditioning long-term development outcomes can be aided through the formation and analysis of scenarios (*23*). The use of input−output (IO) based models to analyze scenarios about actions that could be taken to achieve environmental and social objectives date back to Leontief's use of the World Model (*24*) which, as Duchin (*25*) points out in her review, was an "innovative" framework which did not provide unique, optimal solutions, but rather captured the most "critical attributes" for analyzing data without constraining them. Like all input−output models, the interdependencies among production, consumption, and trade are maintained, providing a powerful framework for exploring economy-wide repercussions of technology change. The coupling of environmental externality data to IO models provides a means to assess the impacts of new technologies while assuring consistency among projections in production, consumption, and the environmental repercussions.

Duchin (*26*) herself utilizes a closed input−output based linear programming model of global trade known as the World Trade Model (WTM) to explore scenarios about actions that could be taken to move toward more sustainable development through the optimization of global trade flows based on regional comparative advantages in production. Juliá and Duchin (*27*) expand the WTM's application to explore potential impacts of climate change scenarios on food production, and Strømman et al. (*28*) examine the relationship between global caps on CO_2 emissions and location of production. Verberg et al. (*29*) make use of an IO framework in their multimodeling approach for exploring development scenarios to assess impacts of future land-use changes in Europe, while Faber et al. (*30*) and Wilting et al. (*31*) employ environmentally extended IO (EE-IO) for assessing envi-

* Corresponding author e-mail: ryan.m.bright@ntnu.no; phone: +47 73 59 89 72; fax: +47 73 59 89 43.

ronmental implications of scenario-specific technological developments of the Dutch economy at the sectoral level. For a comprehensive discussion on the benefits of IO-based modeling, see ref *28*.

Objectives. We explore scenarios about forest-biofuel production and consumption throughout the Fenno-Scandinavian region employing a multiregional, EE-IO framework to assess the climate mitigation opportunities over the medium- and long-term time horizons. Building on previous work (*8*) where GHG mitigation benefits were expressed as a factor of the carbon-intensity of liquid fuels supplied to the Norwegian economy through 2050, we now expand the scope to cover the entire region while paying special attention to evolving consumption patterns in the road transport sector. Our purpose here is to assess, in greater detail, explicit factors which may hinder or enhance the mitigative effectivness linked to the supply of second generation biofuels. In other words, we address the main question: How might the size of the GHG mitigation wedge (*32*) attributed to the supply of biofuels be shaped given assumptions about developments in end-use energy efficiencies (vehicle technologies) and affluence (consumption volume and structure) in road-based transportation? We develop our model so that scenario parameters are linked to the *I* = PAT analogy (*33*) to quantify the mitigation potential of a given biofuel supply under varying assumptions surrounding future developments in road-transport consumption structure, volume, and technology. To keep the scenario analysis simple and transparent, scenarios are developed and presented with explicit assumptions, portraying storylines of potential futures around which decision-makers and stakeholders can begin discussions about the viability of reaching targets, and the necessary trajectories and challenges for developing and implementing technologies and policy.

We start by estimating country-specific forest-biofuel production potentials and likely technology deployment paths to devise three regional fuel-mix scenarios regarding the rates and quantities of forest-biofuels forecasted to replace conventional fossils from 2020 to 2050 in the Fenno-Scandianavian region. We then analyze country-specific trends in consumption patterns—both growth in demand volume and change in demand structure—for developing two regional consumption scenarios, and in addition, two regional scenarios regarding the deployment of more fuel-efficient vehicle technologies up to 2050. A total of four alternative regional scenarios are analyzed and benchmarked to a baseline reference scenario to quantify GHG mitigation potentials obtainable by forest-biofuel based road transport throughout the region. Results of our scenario analyses are used to frame a policy discussion surrounding the implications of forest-biofuels as a medium- and long-term climate strategy in regional road transportation.

Methods and Data

We employ a mixed-unit multiregion input−output (MRIO) model inclusive of global production and consumption activity characterized by year 2000 technology structure and trade flows and extended with three global warming emissions: CO_2, CH_4, and N_2O. Input−output tables are from Eurostat (*34*) and modified GTAP 6 (*35*) provided by Peters and Hertwich (*36*). Air emissions are from WRI (*37*) and Eurostat (*38*). In general, impacts can be expressed mathematically in EE-IO as

$$\mathbf{f} = \mathbf{CF}(\mathbf{I} - \mathbf{A})^{-1}\mathbf{y}$$

where **y** is a vector of the final demand purchases from each economic sector, **A** is a matrix showing the monetary (and physical) relationship between different regions and their

sectors in the global economy (the technology), **I** is the identity matrix, **F** is a matrix with the rows representing the emission intensities of each pollutant in each sector, **C** is a matrix of characterized impact factors, and **f** is a vector with each element representing the resulting impacts for a given final demand. In our model, a wood-based biofuel industry is added comprising 37 detailed mixed-unit processes in each of the three focal countries. By mixed-unit we mean that a combination of physical and economic data is used to characterize the technology structure, such as, for example, inputs of wood and fuel expressed as m^3 and MJ, and inputs of machinery as €. Two conversion technologies representing Fischer−Tropsch diesel (FTD) production and one representing thermochemical ethanol production comprise the new biofuel sector. These and other anciliary processes in the biofuel producing system are described in the Supporting Information and in our previous work (*8, 9*). The share of road-based transport of each country's land transport sector is disaggregated into three mixed-unit passenger transport processes and one goods transport process: *LDV Private Car, LDV Public Unscheduled Taxi, HDV Scheduled Bus, HDV Truck Freight*. Additionally, domestic refining sectors of each region are disaggregated with conventional gasoline and diesel converted into physical units.

Model Parameters. Our parameters follow the *I = PAT* analogy where the impacts (*I*) from road transportation in each country of our focal region are the product of its population (*P*), affluence (*A*), and technology (*T*) (*33*), with *Impact* being analogous to **f** in the above EE-IO equation. Changes in fuel type and use intensity are analogous to *Technology* evolution, or changes to fuel input coefficients within the technology matrix (**A**), while consumption volume (passenger-km, tonne-km; pkm, tkm) and structure (passengers/vehicle; p/v) describe the level of a population's *Affluence* (*PA*) with respect to road transportation activity (**A, Y**). These parameters are defined exogenously based on literature sources. Assumptions regarding annual growth rates in the volume of consumption for each mode in addition to projected deployment rates of more fuel-efficient technologies in each country in our focal region are based on PRIMES (*11, 12*), a partial-equilibrium model of the EU energy system. In our model, consumption volume is expressed in physical units and is the total output for road-based transport including intermediate consumption by industry plus net final demand (i.e., **y** − households, government expenditure, gross capital formation, and exports). Consumption structure parameters describe how the four road transport processes are being consumed, such as, for example, the amount of car-sharing and private vehicle ownership per capita—and are based on country-specific trend extrapolation using data provided by national statistical agencies and other research institutions (*13–17, 35*). Trend data for *LDV Taxi* and *HDV Bus* ridership structure in Norway are used as proxies for the entire region. Consumption structure is another indicator of affluence exemplified as follows: 1 passenger operating a private vehicle over 1 km (vehicle kilometer; vkm) equates to a demand of 1 pkm and requires vehicle operation over 1 km, or 1 vkm/pkm, while 2 passengers/vehicle (2p/v) requires inputs of 0.5 vkm/passenger to fulfill the same demand requirement of 1 pkm, thus as car-pooling increases or decreases over time, the structure of consumption evolves such that the total volume required is affected.

Regional scenarios involving changing fuel use intensities, final demands, and consumption structure comprise what are henceforth labled *Consumption Scenarios*, and regional scenarios involving forest-based biofuel substitution comprise our *Fuel Mix Scenarios*. In total, four alternative production−consumption scenarios are assessed with results benchmarked to a *Baseline Reference Scenario*. This baseline reflects current trends in technical progress, public behavior,

% change	BAU consumption scenario					Slow Growth consumption scenario				
	2001−10	2011−20	2021−30	2031−40	2041−50	2001−10	2011−20	2021−30	2031−40	2041−50
GDP	2.6	2.2	1.7	1.2	0.7	2.0	1.7	1.3	0.9	0.6
HDV freight (Mtkm)	2.4	1.9	1.6	1.1	0.5	1.8	1.4	1.2	0.8	0.4
HDV bus (Mpkm)	−0.1	0.3	0.3	0.3	0.3	0.1	0.3	0.3	0.3	0.4
LDV taxi (Mpkm)	0.0	0.1	0.1	0.1	0.2	0.0	0.1	0.1	0.2	0.2
LDV private (Mpkm)	2.3	1.2	0.8	0.5	0.3	1.8	0.9	0.6	0.4	0.2
energy intensity (TJ/Mpkm)	−0.5	−0.9	−0.9	−0.8	−0.7	−0.8	−1.4	−1.3	−1.2	−1.1
energy intensity (TJ/Mtkm)	0.7	−0.1	−0.4	−0.5	−0.5	0.3	−0.3	−0.6	−0.7	−0.8
p/v, private	−0.1	−0.1	−0.1	−0.1	−0.1	0.1	0.2	0.3	0.4	0.5
p/v, taxi	0.0	0.0	0.0	0.0	0.0	0.1	0.2	0.3	0.4	0.5
vkm/t, freight	0.2	0.2	0.2	0.2	0.2	−0.2	−0.4	−0.5	−0.8	−1.2
p/v, bus	0.0	0.0	0.0	0.0	0.0	0.1	0.2	0.3	0.4	0.5

[a] Mtkm = Million tonne-kilometers; Mpkm = Million passenger kilometers; TJ = Terajoule; p/v = number of passengers/vehicle; vkm/t = vehicle kilometers/tonne goods transported as freight.

energy markets, and regulatory policies, assuming that these trends will basically continue in the future.

Consumption Scenarios. The premise of our *Baseline Reference* consumption scenario follows a "business-as-usual" (BAU) storyline where projections of final demands, structural characteristics, and changing fuel intensities are based on a combination of country-specific data from refs *11* and *12* and by extrapolating statistical trends in each country using data provided by national and European statistical bureaus (*13–17, 39*). This consumption scenario—hereafter referred to as *BAU Consumption*—is not a worst case scenario; rather, it represents a scenario for how the road transport picture may look over the next four decades given today's climate and energy policies. Table 1 presents a weighted-average parametric evolution for the *BAU Consumption* scenario in annual changes, grouped by decade.

Nonroad transport final demand for goods and services are scaled to GDP projections (*11, 12*) for each country. Energy intensities are requirements of the various energy inputs in the fuel mix per unit transport output. Emission factors for each type of energy input are used to derive emission intensities for each transport process, with total direct emissions as the product of emission intensities and total induced outputs for each transport process. In our transport process models, input requirements and fuel-dependent fossil emission intensities are adapted and hybridized using data from ref *40*, and biofuel emission intensities (tank-to-wheel) are adapted from refs *41* and *42*. We model *Private LDV* transport as a separate process with no intermediate consumption, only final consumption ($\mathbf{y}$).

A second, *Slow Growth* consumption scenario is built with a storyline centered on the premise that aggressive regional carbon policy developments together with a reduced oil supply contribute to high energy prices and thus constrain economic activity, resulting in lower rates of economic growth including lowered demands for the various road transport modes (affecting consumption volume), increased public transport occupancy rates and car-pooling (consumption structure), and an accelerated deployment of more energy-efficient vehicle technologies (technology change affecting fuel use intensity). Annual growth rates in final demand (volume) are reduced for private and freight transport and increased for public transport (taxi, bus) under this scenario. In addition, accelerated deployment of more energy-efficient vehicle technologies occur under this scenario and resemble the ACT Map scenario developed by the IEA (*40*). Occupancy rates in public and private transport as well as reductions in the average freight distances traveled per tonne goods (vkm/tonne) requiring transport are modestly improved under this scenario. Regional developments in all parameters of the

Slower Growth consumption scenario are shown on the right side of Table 1.

Fuel Mix Scenarios. Creating our *Fuel Mix* scenarios first involved an investigation into the surplus forest bioenergy potentials and viable biofuel production technology deployment paths at the country-specific level. The sources of biofuel feedstock include forestry and wood processing industry byproduct streams as well as primary timber. The fraction of primary timber comes only from the volume of surplus annual forest growth increment, which is currently growing faster than wood industry output in the region. Figures for future availability account for past trends and industry projections compiled utilizing a variety of national statistical data and literature estimates (*10, 43–47*).

We consider centralized FTD and thermochemical (TC) ethanol production in each country, along with integrated pulp mill FTD production based on black liquor gasification (BLG) in Sweden and Finland only. More information about the sources of forest-derived feedstocks considered, along with information about system designs for FTD and TC-ethanol production, can be found in our previous work surrounding forest-based biofuels in Norway (*8, 9*). We base our integrated FTD design around CHEMREC's BLG-DME process, adapting process inventories from ref *48* and scaling inputs and emissions according to BLG-FTD yields (*49, 50*). Upstream biomass transport processes of our Well-to-Tank inventories for BLG-FTD cases in Sweden and Finland are adjusted following information obtained in refs *51* and *52*. Country-specific annual surplus bioenergy potentials are shown in Table 2 along with biomass-biofuel conversion efficiencies. Because additional wood inputs are required as black liquor substitutes to meet plant heat and steam demands, we label this biomass fraction "black liquor substitute" in Table 2 and account for it in our resource assessment. For the integrated production system, efficiencies are based on a partitioning of the mill's wood inputs to the fraction required by the BLG plant. We assume constant production efficiencies across all years and scenarios because our knowledge is limited as to what would be reliable technical improvements to expect and at what time they would occur.

In total, three fuel mix scenarios are developed: a *BAU Fuel Mix* scenario, which comprises part of our *Baseline Reference Scenario*, along with two high biofuel mix scenarios. The high biofuel mix scenarios are created to assess the variations between production-based and consumption-based biofuel targets. In the reference scenario, due to the likelihood that second generation biofuels will inherently comprise part of the transport fuel mix over the medium- and long-term horizons, we assume modest deployment rates

TABLE 2. Biomass Input Ratios, Annual Total Bioenergy Potentials, Biomass Conversion Efficiencies, and Unit Level Life Cycle Global Warming Impacts of the "Well-to-Tank" Biofuel and Reference Fuel Production Systems[a]

biomass input share	Norway	Finland	Sweden
primary forest biomass	69%	30%	44%
logging/secondary industry residuals	31%	27%	32%
black liquor substitute	0%	43%	26%
total annual potential, PJ, 2020−2050	119	284	428

conversion efficiency (LHV basis)	all countries		
centralized TC E100	46%		
centralized FTD100	45%		
BLG-FTD	43%		

well-to-tank GWP (tonne-CO$_2$-eq./TJ)	Norway	Finland	Sweden
conventional gasoline	9.3	10.8	10.1
conventional diesel	10.4	11.2	11.6
TC E100	15.8	15.2	14.9
FTD100	16.5	16.6	16.0
BLG-FTD100	N/A	12.6	12.3

[a] Life cycle inventory of TC E100 production is compiled using data from ref 53; FTD100 inventory data are adapted from refs 53 and 54; and BLG-FTD100 is compiled using data adapted from refs 48–50. Conversion efficiencies (49, 53, 54) are calculated on a lower heating value (LHV) basis and 21.5 MJ/kg DM for all biomass types. Lifecycle Impact Assessment is performed using the CML Baseline 2000 Impact Method, 100-year global warming equivalents (55); GWP = Global Warming Potential; TC = Thermochemical; TJ = Terajoule; FTD = Fischer−Tropsch Diesel; BLG = Black Liquor Gasification.

and adopt IEA projections of second generation diesel and ethanol contributions in the global transport fuel mix as proxies. Second generation diesel and ethanol shares are 1% and 1% at 2020, respectively, and 20% and 5%, respectively, at 2050 (56). Since we are only concerning ourselves with second generation forest-biofuels, we exclude all others and begin substituting conventional fuel with biofuel starting in 2020. We assume no biofuels are in the transport fuel mix at base year 2000. In the first alternative biofuel mix scenario—High Biofuel Energy Share—biofuel substitutes fossil fuel on an end-use energy demand share basis for all road transport processes following EU targets of 10% at 2020 (2), reaching 50% at 2050. In a second, alternative biofuel mix scenario—High Biofuel Output—we maintain the same biofuel input requirements as the High Biofuel Energy Share scenario and total fuel input requirements required under the Slow Growth Consumption scenario for each mode, with the share of fossil requirements comprising the difference. The result is a weighted average biofuel (end-use energy) share that approaches 90% at 2050 (Figure 1). The share of conventional diesel relative to gasoline in all scenarios remains fixed and is based on trend extrapolation of country-specific fuel demands in road transport (14).

Well-to-Wheel Scenarios. Table 3 presents a descriptive overview of how our two Consumption scenarios (BAU, SG) couple with our three Fuel Mix scenarios.

In total, four alternative production-consumption scenarios are assessed with results benchmarked to a Baseline Reference Scenario which pairs BAU Consumption with the BAU Fuel Mix scenario. Scenario 1 joins BAU Consumption

with the High Biofuel Energy Share fuel mix scenario; Scenario 2 couples the SG Consumption with the BAU fuel mix scenario; Scenario 3 couples SG Consumption with High Biofuel Energy Share; and Scenario 4 represents a well-to-wheel scenario that joins SG Consumption together with the High Biofuel Output fuel mix scenario. Underlying assumptions in the input data, methods, and scenarios should be kept in mind when interpreting the results.

Results

If BAU developments in consumption, forest-biofuel, and efficient vehicle technology deployment up to 2050 are realized, annual direct GHG emissions from fuel production and road transport activity will increase from today's levels (2008) by 40% at 2050, mostly due to growth in private and freight transport. Total fuel consumption for regional road transport over the 30-year period of the Baseline Reference Scenario is 24.1 exajoules (EJ), of which 2.7 EJ (57) is forest-biofuels. Meeting a 25% biofuel target (Figure 1) at 2050 under this scenario would require the utilization of 53% of the regional resource base, represented by the cumulative shaded area below the black line labeled "BR" in Figure 1. Total cumulative GHG emissions associated with fuel production and road transportation are around 1 Gt-CO$_2$-eq.

A more aggressive biofuel infusion policy like that of Scenario 1 ("S1", Figure 2) where biofuels comprise a 50% share of all transport fuel consumed by regional road transport at 2050 would yield net annual avoided emissions of 16 Mt-CO$_2$-eq. at 2050, reductions stemming mostly from private and freight transport. Total biofuel consumption increases to 7.1 EJ over the 30-year period which mitigates 182 Mt-GHGs—a reduction of 19% from the Baseline—the integral of the dark wedge, top left of Figure 2. However, reaching a 50% biofuel target would require either imports of biomass feedstock from outside the region or use of nonforest-derived feedstocks beginning around 2046 as shown in Figure 1. Additionally, a return to 2000 GHG emission levels (dashed line, Figure 2) cannot be met with a stand alone biofuel policy, and GHG emissions from biofuel production offset some of the benefits of biofuel consumption in road transport.

In Scenario 2, maintaining the same BAU fuel mix scenario as that of the reference scenario when incorporating the parameter changes of the Slow Growth consumption scenario results in a cumulative mitigation potential of 403 Mt-CO$_2$-eq.-avoided. This is a 121% increase over the mitigation potential of Scenario 1 and implies that an altered consumption profile combined with earlier deployment of more efficient vehicle technologies can be significantly more effective than increasing the supply of biofuel in the fuel mix. Under such a scenario, a return to 2000 emission levels can be realized around 2037. Total fuel requirements in road transport are reduced from 24.1 to 14.5 EJ, of which forest-biofuels comprise 1.5 EJ. Additionally, the resource utilization rate under this scenario (Figure 1) is drastically reduced—only 24% of the surplus resource base is required to meet a 25% end-use biofuel share at 2050. Further, less total biofuel production is required to meet the same share targets as the reference scenario resulting in additional mitigation benefits.

The effect of combining the more aggressive biofuel mix scenario of S1 with the alternative consumption scenario of S2 results in additional GHG savings of 101 Mt-CO$_2$-eq.-avoided for Scenario 3 (bottom left, Figure 2) over S2; however, the increase comes at the expense of significantly higher levels of resource use (shown in Figure 1). Although both Scenarios 2 and 3 involve the Slow Growth consumption parameters, increasing the amount of biofuel from 1.5 to 3.9 EJ in the fuel mix under Scenario 3 more than doubles the
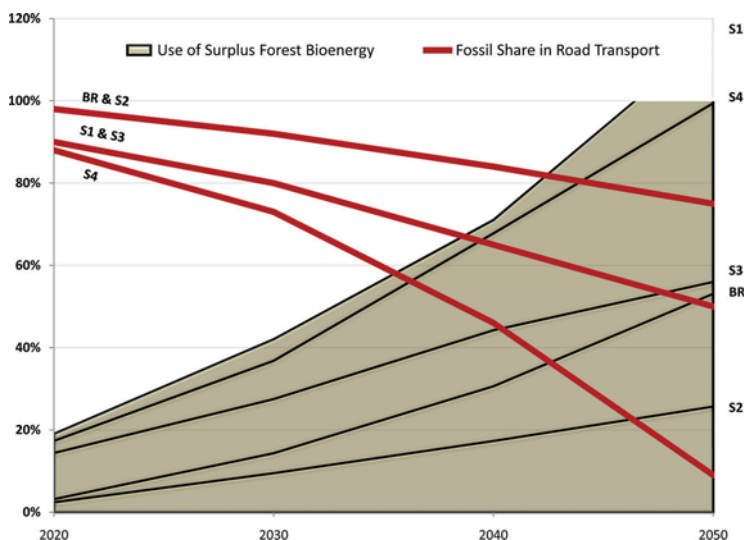
FIGURE 1. Annual changes in the (energetic) share of conventional fossil fuel in the transport mix (red lines) and utilization of the region's projected annual surplus forest bioenergy supply under each scenario (black lines). BR = *Baseline Reference*; S1 = *Scenario 1*; S2 = *Scenario 2*; S3 = *Scenario 3*; S4 = *Scenario 4*.

TABLE 3. Scenario Descriptions (BR = *Baseline Reference*; S1 = *Scenario 1*; S2 = *Scenario 2*; S3 = *Scenario 3*; S4 = *Scenario 4*)

n	Name	"P · A" - Consumption	"T" - Fuel Efficiency	"T" - Fuel Mix	Purpose
BR	Baseline Reference	- BAU demand growth - BAU structural change	- BAU tech. deployment	- BAU biofuel deployment	- Reference - Not worst case
S1	BAU Consumption + High Biofuel, Consumption-based Targets	- BAU demand growth - BAU structure	- BAU deployment	- High biofuel *energy share targets* following EU directive	- Emission trajectory resulting from a stand-alone biofuel, share-based policy
S2	Slow Growth (SG) Consumption + BAU Fuel Mix	- Carbon policy slows demand growth rates - Structural changes: Less private vehicle ownership/capita; higher vehicle occupancy rates	- Accelerated deployment of more fuel efficient vehicle technologies	- BAU biofuel deployment	- Emission trajectory resulting from aggressive efficiency and consumption-oriented policy
S3	SG Consumption + High Biofuel, Consumption-based Targets	- Same as **S2**	- Same as **S2**	- Same as **S1**	- Emission trajectory resulting from aggressive consumption and biofuel policies
S4	SG Consumption + High Biofuel, Production-based Targets	- Same as **S2**	- Same as **S2**	-High *biofuel fixed output targets* (Maximum self-sufficient production)	- Emission trajectory resulting from aggressive consumption and biofuel policies (production-oriented biofuel targets)

resource utilization rate yet only reduces GHG emissions by an additional 25%. However, 2000 emission levels can be obtained at around 2026—about a decade earlier under this scenario.

The rationale behind creating the final alternative scenario—*Scenario 4*—is to assess GHG benefits of a policy scenario involving production-based as opposed to consumption-based biofuel targets. Referring to Figure 1, we find that the share of conventional fossil fuel in the fuel mix is greatly reduced under this scenario, and as the new regional output for conventional fuel is lowered, so too does the total output for all road transport processes due to Leontief multiplier effects. This in turn lowers the total

output of biofuel to a level where near self-sufficient biofuel production can be sustained in the region at 2050 (Figure 1), where 100% utilization of the resource base displaces 90% of all fossil fuel consumption in road transport. The implication of this finding is that a shift in consumption patterns increases the relative share of biofuel in the total fuel mix which lowers the carbon intensity and expands the mitigation wedge. Under this scenario, 6.1 EJ biofuel is consumed, and cumulative net emission reductions are 604 Mt-GHGs-avoided—a 232% increase in the mitigation effectiveness linked to the biofuel supply over *Scenario 1*. Additionally, 2000 emission levels can be reached around 2022.
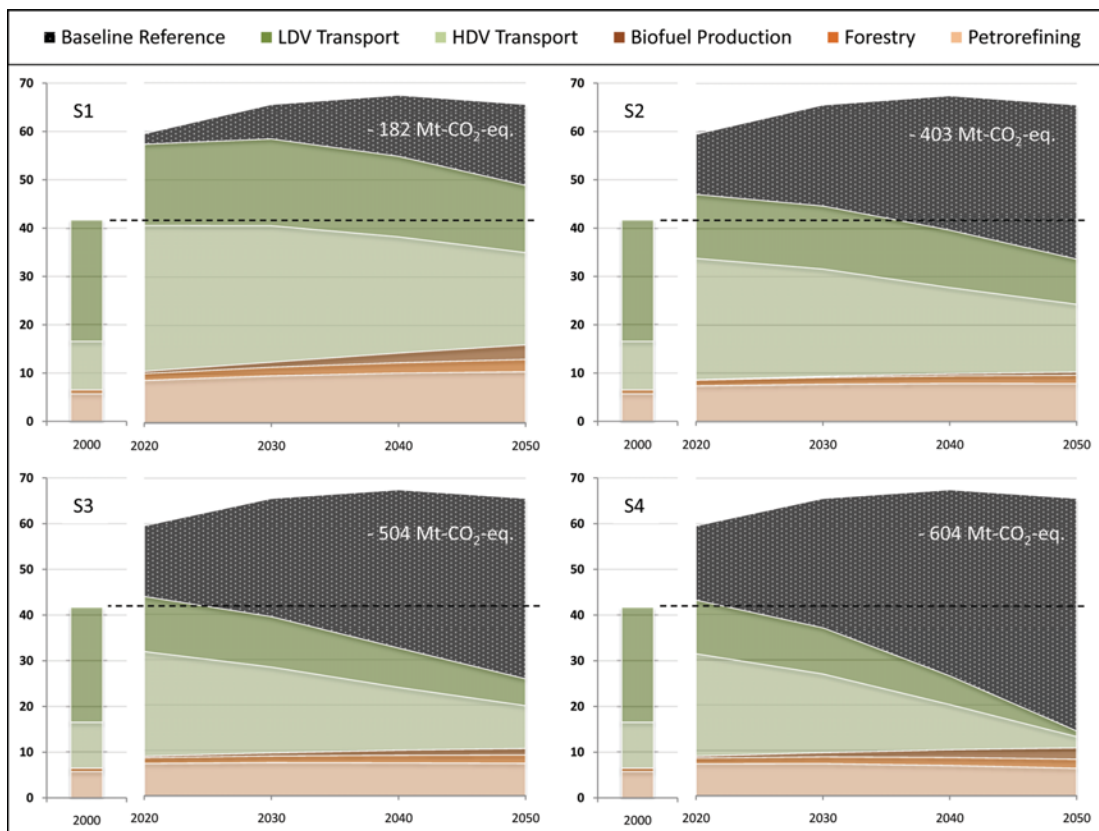
FIGURE 2. Year 2000 GHG emission in the region from fuel production, forestry, and road transportation (left bar) shown with emission trajectories of the *Baseline Reference* (black shaded area) and the four alternative scenarios (*y*-axis = Million tonnes CO$_2$-equivalents). Passenger bus + freight transport = *HDV Transport*; Taxi + Private transport = *LDV Transport*; Dashed black line = year 2000 emission reference.

Discussion

We illustrate that a combination of lowered final demand volumes (Mt-, pkm), altered final demand structures (p/v, vkm/t), and more efficient vehicle technologies (TJ/Mt-, pkm) can lead to a reduction in total transport fuel consumption, in turn enhancing the mitigation effectiveness of biofuels as their relative contribution in the total fuel mix increases. In consumption scenarios built around such a strategy as in *Scenario 2*, which coupled *Slow Growth* consumption with "business-as-usual" biofuel deployment (*BAU Fuel Mix* scenario), around 121% more GHGs could be avoided over a scenario involving more aggressive biofuel targets coupled with a "business-as-usual" evolution in consumption parameters (*Scenario 1*). In other words, the carbon-intensity of the energy supply required to meet the region's future road transport demands can be deeply influenced by factors other than stand alone aggressive biofuel policies, either by reducing final demand volumes (total number of t-, pkms) or by reducing the energy intensity per unit demand either through altered demand structure (number vehicle kilometers/passenger kilometer) or technical improvements (fuel efficiency). However, we recognize the existence of some intrinsic uncertainty associated with the results of our scenarios involving modified consumption parameters. The likelihood with which the *Slow Growth* consumption scenario's parameters can evolve simultaneously at the rates modeled in this study over the medium- and long-term horizons should be kept in mind. Nevertheless, we should reiterate that our objective was to keep the scenario analysis

simple and transparent, creating scenarios with explicit assumptions to portray storylines of potential futures around which decision-makers can begin discussions about the viability of reaching the mitigation targets and the necessary trajectories and challenges associated with developing and implementing future technologies and policy.

A limitation of our study was our decision to exclude the long-term effects of technological improvements as it is likely that biofuel production efficiencies may gradually improve over time through learning by doing, as evidenced by the Brazilian sugar cane and U.S. corn ethanol industries. To test the sensitivity of this parameter, the production efficiencies shown in Table 2 were increased 5% for our three conversion technologies in our *Baseline Reference Scenario* which led to additional savings of 60,000 tonnes GHGs at 2050 and cumulative savings of 860,000 tonnes for the entire 30-year period.

Reducing transport volumes throughout the region will require difficult and sustained efforts on all fronts, and over the short- and medium-terms, reductions in transport volumes for the purpose of mitigating GHGs without jeopardizing economic growth will be challenging to obtain (*58*). Yet volume-reducing measures would ease the difficulty and costs of achieving substantial GHG reductions (*59*), and furthermore, achieving long-term emission cuts within the road transport sector to the required 2050 stabilization levels would likely be difficult in their absence (*59, 60*). In the short- and medium-terms, volume-reducing policies would probably require some combination of fiscal policy measures such

as elevated carbon and mileage taxes (22), increased tolling, congestion taxes (21, 61), road pricing (62), subsidized public transport, and sales/circulation taxes aimed at reducing private vehicle ownership/capita (63). Over the long-term, optimized spatial organization and physical planning may begin to play a greater role at reducing transport volumes (58).

Less uncertain in our modeling are the rates at which more efficient LDV and HDV transport technologies are deployed under our *Slow Growth* consumption scenario, as generally the cost-effectiveness potentials associated with "on-road" and vehicle efficiency improvements are significant (58, 59), and given higher medium- and long-term carbon prices, such "no-regrets" (59) measures are likely to be exploited before other measures. For the latter instance to be environmentally effective, however, lowered fuel consumption through improvements in vehicle technology must not lead to increased consumption volumes. This direct rebound effect—sometimes referred to as Jevons Paradox (64)—implies that policy measures targeting efficiency improvements must also be designed in a way that new demands do not arise as a result (63, 65, 66). Brännlund et al. (67) showed that efficiency improvements in Swedish road transport reduced fuel costs per mile such that demands for private transport increased resulting in a positive rebound effect. Greening et al. (65) reviewed 22 studies and report a 2−6% combined rebound effect (direct, indirect, and macroeconomic) associated with 20% efficiency gains in automotive transport. Thus complementary policies which help to ensure demand inelasticities such as, for example, by setting a floor on transport fuel prices via energy taxes, should be considered in conjunction with those targeting energy efficiency improvements.

Policies targeting changes in consumption structure can lead to reductions in energy intensities per passenger- or tonne-kilometer and include measures aimed at increasing occupancy rates, reducing fleet average car/engine sizes, reducing congestion (through car-pooling, infrastructure modifications such as through the provision of more high-occupancy vehicle lanes) (20) and reducing high-speed driving. Other energy-intensity reduction measures promoting overall vehicle efficiency improvements include drag reduction, auxiliary component energy consumption (i.e., air conditioning, power electronics, etc.), improved tire pressure, and requirements of lightweight material composites.

While we capture the benefits of less fuel-intensive road transportation, we do not explicitly model the use of the region's biomass resources for increased production of nonliquid energy carriers such as biopower, a viable transport fuel capable of contributing to alternative road-based transport, particulary LDV transport. The use of bioelectricity as an energy carrier has been shown to be a more efficient use of land and bioresources for achieving alternative road transport objectives than biofuels like ethanol (68, 69). Technologies such as plug-in hybrid electric vehicles which can use both biofuel and biopower exist in near-commercial form, and biopower can be obtained from cogenerated heat and power (CHP) or electric-only power stations (IGCC)—technologies which exist in fully commercial, economically viable form. As Ohrogge et al. point out, although there are uncertainties in the pace of electric car development and market penetration, future strategies aimed at promoting bioelectricity instead of ethanol for substituting conventional fuels like gasoline in cars and promoting more diesel engines in heavier vehicles may be the best route to the goal of reducing petroleum consumption and CO_2 emissions (69).

There are several potentially adverse implications that a scaled up forest-biofuels program could have on the region's global warming mitigation potential that should be considered prior to implementation of new or modified forest-biofuel support policy. Ecosystem changes as a result of changing land use through more intensive forestry may impact feedbacks to climate change which involve land ecosystem—atmosphere interactions (70, 71). These land ecosystem changes may affect biophysical factors such as surface albedo, surface roughness, and the surface moisture budget which could alter local and regional temperatures and should be examined carefully alongside changing forest carbon budgets. Expanded or more intensified forestry may also affect biogeochemical interactions in ways negatively affecting the region's long-term mitigation potential, thus other important considerations such as the nutrient economy of the forests including the various options of nutrient generation, recycling and fertilizer compensation, soil emissions, and carbon and nitrogen cycles, should also be considered. For more discussion on these aspects and others see ref 8.

The results of our study clearly illustrate that, although there is significant mitigation potential through replacing fossil fuel with forest-biofuels, stand-alone biofuel policies fall far short of realizing the GHG mitigation potential that exists in the way of reducing consumption volume, altering consumption structure, and increasing energy efficiency. Complementary policies oriented around the three latter strategies have the potential to significantly leverage the mitigation effectiveness linked to any future supply of biofuels and facilitate a transition away from a fossil-dependent road transport sector for reducing the region's carbon footprint. Further, for reasons extending beyond climate change mitigation, a combination of biofuel and effective consumption policies can lead to transport that is truly sustainable, as the results of *Scenario 4* show that the resource base can contribute significantly to regional self-sufficient biofuel production and consumption, enhancing energy security benefits. However, self-sufficient and sustainable road-based transport can only be made possible through continued efforts at reducing impacts from consumption and promoting more fuel-efficient vehicles. A portfolio of complementary strategies aimed at maximizing the efficiency with which regional bioenergy resources are used to fuel alternative transport are surely required.

Acknowledgments

Supporting Information Available

Additional methodological descriptions, tables showing the processes which comprise our biofuel sector, and the types and amounts of biofuel replacing conventional fossil fuel under each scenario. This material is available free of charge via the Internet at http://pubs.acs.org.

Literature Cited

(1) European Environment Agency. *Greenhouse Gas Emission Trends and Projections in Europe 2008: Tracking Progress Towards Kyoto Targets*; Report 5/2008; European Environment Agency (EEA): Copenhagen, 2008; accessed May 25, 2009 at http://www.eea.europa.eu.

(2) European Commission. *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 and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC*; Brussels, 2009.

(3) Hill, J.; Polasky, J.; Nelson, E.; Tilman, D.; Huo, H.; Ludwig, L.; Neumann, J.; Zheng, H.; Bonta, D. Climate change and health costs of air emissions from biofuels and gasoline. *Proc. Natl. Acad. Sci.* **2009**, *106* (6), 2077–2082.

(4) Koh, L. P.; Ghazoul, J. Biofuels, biodiversity, and people: Understanding the conflicts and finding opportunities. *Bio. Cons.* **2008**, *141*, 2450–2460.

(5) OECD. *Biofuel Support Policies: An Economic Assessment*; Organization for Economic Cooperation and Development: Paris, 2008.

(6) Gallagher, E. *The Gallagher Review of the Indirect Effects of Biofuels Production*; Renewable Fuels Agency (RFA): East Sussex, UK, July, 2008.

(7) IEA/OECD. *From 1st- to 2nd-Generation Biofuel Technologies: An Overview of Current Industry and RD&D Activities*; IEA, OECD: Paris, November, 2008.

(8) Bright, R. M.; Strømman, A. H.; Hawkins, T. R. Environmental assessment of wood-based biofuel production and consumption scenarios in Norway. *J. of Ind. Ecol.* **2010**, *14* (3), in press.

(9) Bright, R. M.; Strømman, A. H. Life cycle assessment of second generation bio-ethanol produced from Scandinavian boreal forest resources: A regional analysis for middle Norway. *J. Ind. Ecol.* **2009**, *13* (3), 514–531.

(10) Econ Pöyry. *The expanding bio-energy market in the Nordic countries—possibilities and consequences*; Report No. 2007-091; Econ Pöyry AS, Commissioned by Nordic Energy Research: Copenhagen, 2008; accessed December 18, 2008 at http://www.nordicenergy.net.

(11) Mantzos, L.; Capros, P. *European energy and transport: Trends up to 2030—an update*; Institute of Communication and Computer Systems of the National Technical University of Athens—Prepared for the Directorate-General for Energy and Transport, European Commission: Belgium, May 22, 2006.

(12) Capros, P.; Mantzos, L.; Papandreou, V.; Tasios, N. *European energy and transport: Trends to 2030— update 2007*; Institute of Communication and Computer Systems of the National Technical University of Athens—Prepared for the Directorate-General for Energy and Transport, European Commission: Belgium, April 8, 2008.

(13) Eurostat. *EU Energy and Transport in Figures: Statistical pocketbook 2007/2008;* Office for Official Publications of the European Communities: Luxembourg, 2008; accessed March 27, 2009 at http://ec.europa.eu/dgs/energy_transport/figures/index_en.htm.

(14) Eurostat. *Panorama of Transport: Edition 2007*; Eurostat, European Commission: Office for Official Publications of the European Communities: Luxembourg, 2007.

(15) Finish Road Administration. *Finnish Road Transport Statistics Online Database*; Finish Road Administration (TIEHALLINTO), 2009; accessed February 24, 2009 at http://www.tiehallinto.fi.

(16) TØI. *Transportytelser i Norge 1946—2007 [Transport volumes in Norway 1946—2007]*, Report 979/2008; Institute of Transport Economics (TØI): Oslo, September, 2008.

(17) SIKA. *Persontransporternas utveckling till 2010 [Passenger transport development until 2010];* Swedish Institute for Transport and Communications Analysis (SIKA), 2002; accessed February 12, 2009 at: http://www.sika-institute.se.

(18) SIKA. *Körsträckor år 2006 [Distances covered in 2006]*; Swedish Institute for Communications Analysis (SIKA), 2007; accessed March 15, 2009 at http://www.sika-institute.se.

(19) Tapio, P. Towards a theory of decoupling: degrees of decoupling in the EU and the case of road traffic in Finland between 1970 and 2001. *Trans. Policy* **2005**, *12* (2), 137–151.

(20) Harrington, W.; Krupnick, A. J.; Alberini, A. Overcoming public aversion to congestion pricing. *Trans. Res. Part A: Policy Practice* **2001**, *35* (2), 87–105.

(21) Parry, I. W. H. Comparing the efficiency of alternative policies for reducing traffic congestion. *J. Public Econ.* **2002**, *85* (3), 333–362.

(22) Parry, I. W. H. Are the costs of reducing greenhouse gases from passenger vehicles negative? *J. Urban Econ.* **2007**, *62* (2), 273–293.

(23) Hojer, M.; Ahlroth, S.; Dreborg, K. H.; Ekvall, T.; Finnveden, G.; Hjelm, O.; Hochschorner, E.; Nilsson, M.; Palm, V. Scenarios in selected tools for environmental systems analysis. *J. Clean. Prod.* **2008**, *16* (18), 1958–1970.

(24) Leontief, W.; Carter, A. P.; Petri, P. *The Future of the World Economy*; Oxford University Press: New York, 1977.

(25) Duchin, F. *International Trade: Evolution in Thought and Analysis of Wassily Leontief*; Cambridge University Press: West Nyack, NY, 2004.

(26) Duchin, F. A world trade model based on comparative advantage with *m* regions, *n* goods, and *k* factors. *Econ. Syst. Res.* **2005**, *17* (2), 141–162.

(27) Julià, R.; Duchin, F. World trade as the adjustment mechanism of agriculture to climate change. *Climatic Change* **2007**, *82*, 393–409.

(28) Strømman, A. H.; Hertwich, E. G.; Duchin, F. Shifting trade patterns as a means of reducing global carbon dioxide emissions: a multiobjective analysis. *J. Ind. Ecol.* **2009**, *13* (1), 38–57.

(29) Verburg, P. H.; Eickhout, B.; van Meijl, H. A multi-scale, multi-model approach for analyzing the future dynamics of European land use. *Ann. Reg. Sci.* **2008**, *42*, 57–77.

(30) Faber, A.; Idenburg, A.; Wilting, H. Exploring techno-economic scenarios in an input-output model. *Futures* **2007**, *39*, 16–37.

(31) Wilting, H. C.; Faber, A.; Idenburg, A. M. Investigating new technologies in a scenario context: description and application of an input-output method. *J. Clean. Prod.* **2008**, *16* (S1), S102–S112.

(32) Pacala, S.; Socolow, R. Stabilization wedges: Solving the climate problem for the next 50 years with current technologies. *Science* **2004**, *305* (5686), 968–972.

(33) Ehrich, P. R.; Holdren, J. P. Impact of population growth. *Science* **1971**, *171* (3977), 1212–1217.

(34) Eurostat: National Accounts Database, 2008; accessed January 19, 2009 at http://epp.eurostat.ec.europa.

(35) Dimaranan, B. V. *Global trade, assistance, and production: The GTAP 6 data base*; Center for Global Trade Analysis, Purdue University, 2006.

(36) Peters, G. P.; Hertwich, E. G. CO_2 embodied in international trade with implications for global climate policy. *Environ. Sci. Technol.* **2008**, *42* (5), 1401–1407.

(37) World Resource Institute. *Earth Trends: The Environmental Information Portal - Climate and Atmosphere Variables*; accessed April 16, 2009 at http://earthtrends.wri.org/text/climate-atmosphere/variables.html.

(38) Eurostat. *NAMEA air emissions by industry (NACE A31) - datasets*; 2008, accessed at http://epp.eurostat.ec.europa.

(39) Eurostat. *Inland freight and passenger transport in the EU-27 up 2007: performances, modal split, and structural indicators*; accessed February 4, 2009 at http://publications.europa.eu.

(40) Spielmann, M.; Bauer, C.; Dones, R.; Tuchschmid, M. *Transport Services—Ecoinvent Report no. 14*; Swiss Centre for Life Cycle Inventories: Dübendorf, Switzerland, 2007.

(41) Röder, A. *Life-Cycle Inventory and Costs of Different Car Powertrains*; Paul Scherrer Institute (PSI): Villigen, Switzerland, December, 2001.

(42) Delucchi, M. A. *A Lifecycle Emissions Model (LEM): Lifecycle Emissions from Transportation Fuels, Motor Vehicles, Transportation Modes, Electricity Use, Heating and Cooking Fuels, and Materials*; Institute of Transportation Studies, University of California: Berkeley/Davis, CA, December 2003.

(43) Statistics Norway. *Statistics by Subject Area—Forests*; accessed February 22, 2009 at http://www.ssb.no/skog_en/.

(44) Statistics Finland. *Forest Accounts*; accessed February 22, 2009 at http://www.stat.fi/til/mettp/index_en.html.

(45) Statistics Sweden. *Forestry Statistics—Current Information about Forest Resources in Sweden*; accessed February 22, 2009 at http://www.scb.se/Pages/Product____9553.aspx.

(46) UN ECE/FAO. *European Timber Trends and Prospects: Into the 21st Century*; Timber Committee of the United Nations Economic Commission for Europe, European Forestry Commission of the FAO, 2000.

(47) FAO. *Forest Resource Assessment (FRA) 2005—Country Tables*; accessed February 22, 2009 at http://www.fao.org/forestry/32103/en/.

(48) Jungbluth, N.; Frischknecht, R.; Faist Emmenegger, M.; Steiner, R.; Tuchschmid, M. Life Cycle Assessment of BTL-Fuel Production: Inventory Analysis 2007; RENEW Integrated Project for Sustainable Energy Systems, Report Deliverable 5.2.7; accessed March 13, 2008 at http://www.renew-fuel.com/fs_documents.php.

(49) Ekbom, T.; Nerglin, N.; Lögdberg, S. *Black liquor gasification with motor fuel production— BLGMF II*; Nykomb Synergetics: Stockholm, Sweden, 2005.

(50) Pettersson, K.; Harvey, S. CO_2 emission balances for different black liquor gasification biorefinery concepts for production of electricity or second-generation liquid biofuels. *Energy* **2010**, *35* (2), 1101−1106.

(51) Eriksson, L. N. Forest-fuel systems—comparative analyses in a life cycle perspective. Doctoral Thesis, Mid Sweden University, Östersund, Sweden, 2008.

(52) González-García, S.; Berg, S.; Feijoo, G.; Moreira, M. T. Comparative environmental assessment of wood transport models: A case study of a Swedish pulp mill. *Sci. Total Environ.* **2009**, *407* (11), 3530–3539.

(53) Phillips, S.; Aden, A.; Jechura, J.; Dayton, D.; Eggeman, T. *Thermochemical Ethanol via Indirect Gasification and Mixed Alcohol Synthesis of Lignocellulosic Biomass*; National Renewable Energy Laboratory (NREL): Golden, CO, April, 2007.

(54) Baitz, M.; Binder, M.; Degen, W.; Deimling, S.; Krinke, S.; Rudloff, M. *Comparative Life-Cycle Assessment for SunDiesel (CHOREN Process) and Conventional Diesel Fuel: Executive Summary*; PE Europe GmbH−Life Cycle Engineering, September, 2004.

(55) Leiden University. *Life Cycle Assessment−An Operational Guide to ISO Standards, Version 2*; Centre of Environmental Science (CML), Leiden University: Leiden, The Netherlands, 2001.

(56) IEA. *Energy Technology Perspectives 2008−Scenarios and Strategies to 2050*; International Energy Agency: Paris, 2008.

(57) Please refer to Supporting Information for a table showing the type and quantity of each biofuel that replaces fossil fuel for each scenario.

(58) Grazi, F.; van den Bergh, J. C. J. M. Spatial organization, transport, and climate change: comparing instruments of spatial planning and policy. *Ecol. Econ.* **2008**, *67* (4), 630–639.

(59) Lutsey, N.; Sperling, D. Greenhouse gas mitigation supply curve for the United States for transport versus other sectors. *Trans. Res. Part D: Trans. Environ.* **2009**, *14* (3), 222–229.

(60) Yang, C.; McCollum, D.; McCarthy, R.; Leighty, W. Meeting an 80% reduction in greenhouse gas emissions from transportation by 2050: A case study in California. *Trans. Res. Part D: Trans. Environ.* **2009**, *14* (3), 147–156.

(61) Verhoef, E.; Nijkamp, P.; Rietveld, P. Tradeable permits: their potential in the regulation of road transport externalities. *Environ. Planning B: Planning Design* **1997**, *24* (4), 527–548.

(62) Kim, K. S.; Hwang, K. An application of road pricing schemes to urban expressways in Seoul. *Cities* **2005**, *22* (1), 43–53.

(63) de Haan, P.; Peters, A.; Scholz, R. W. Reducing energy consumption in road transport through hybrid vehicles: Investigation of rebound effects, and possible effects of tax rebates. *J. Clean. Prod.* **2007**, *15* (11−12), 1076–1084.

(64) Alcott, B. Historical Overview of the Jevons Paradox in the Literature. In *The Jevons Paradox and the Myth of Resource Efficiency Improvements*; Polimeni, J. M., Mayumi, K., Giampietro, M., Alcott, B., Eds.; Earthscan Sterling: Virginia, 2008.

(65) Greening, L. A.; Greene, D. L.; Difiglio, C. Energy efficiency and consumption−the rebound effect−a survey. *Energy Policy* **2000**, *28* (6−7), 389–401.

(66) Gottron, F. Energy efficiency and the rebound effect: does increasing efficiency decrease demand?; Resources, Science, and Industry Divsion, Congressional Research Service (CRS), Library of Congress, 2001; accessed June 8, 2009 at http://www.ncseonline.org.

(67) Brännlund, R.; Ghalwash, T.; Nordström, J. Increased energy efficiency and the rebound effect: effects on consumption and emissions. *Energy Econ.* **2007**, *29* (1), 1–17.

(68) Campbell, J. E.; Lobell, D. B.; Field, C. B. Greater transportation energy and GHG offsets from bioelectricity than ethanol. *Science* **2009**, *324* (5930), 1055–1057.

(69) Ohlrogge, J.; Allen, D.; Berguson, B.; DellaPenna, D.; Shachar-Hill, Y.; Stymne, S. Driving on biomass. *Science* **2009**, *324* (5930), 1019–1020.

(70) Jackson, R. B.; Randerson, J. T.; Canadell, J. G.; Anderson, R. G.; Avissar, R.; Baldocchi, D. D.; Bonan, G. B.; Caldeira, K.; Diffenbaufh, N. S.; Field, C. B. Protecting climate with forests. *Environ. Res. Letters* **2008**, *3*, 044006 (5pp).

(71) Betts, R. Implications of land ecosystem-atmosphere interactions for strategies for climate change adaptation and mitigation. *Tellus B* **2007**, *59* (3), 602–615.

ES903135C

5.6 Uncertainties and Limitations

Apart from the typical uncertainties associated with IO-based modeling frameworks described in Chapter 3, additional uncertainties inherent to MRIO modeling exist. These can be related to sector aggregation discrepancies due to variances in input-output classification across countries, monetary exchange rates/currency conversion, and treatment of the rest-of-world (ROW) region. Further uncertainty stems from the estimation of off-diagonal trade flow matrices, as typically all that is known are vectors from supplying sectors instead of matrices to using sectors. Assumptions are often required to produce these trade flow matrices. In this study, trade flow matrices were estimated using trade coefficients describing the percentage of imports of a particular commodity type into one country that come from another country. As a result, the procedure assumes that the trade coefficients are identical for all entries along a row of the imports matrix (for all using domestic industries) – an assumption likely affecting model accuracy.

The study is limited by the exclusion of other transport-related GHG substances like short-lived chemically active gases such as CO, NO_x, and VOCs (that induce indirect climate effects) as well as aerosols/aerosol precursors like organic and black carbon, for example. Many are short-lived substances, and their global warming impacts are difficult to quantify because the magnitude of their effects vary over time for each transport mode and because impact is very much dependent on the location of emission (Borken-Kleefeld, Berntsen, & Fuglestvedt, 2010; Fuglestvedt et al., 2009). While these substances do have warming impacts over the short-term, they are minimal relative CO_2 in terms of its dominating, long-term climate warming effect (Berntsen & Fuglestvedt, 2008). Additionally, aerosol emission data due to the combustion of ethanol and synthetic diesel made from biomass are difficult to obtain based on current, publically available literature.

5.7 Summary and Conclusions

Despite the inherent modeling uncertainties of MRIO and exclusion of some emission substances, it can be strongly concluded that policies leading to a reduction in overall transport activity and energy intensity are far more effective at reducing emissions than stand alone forest-biofuel policies. Regarding a reduction in total transport activity, it remains to be seen whether this can occur in the face of sustained economic growth due to the tightly coupled nature of road transport and GDP in the region, particularly freight transport (DG TREN, 2009; Tapio, 2005). Aggressive policy targeting reduced energy intensities are needed to compliment the climate benefits of forest biofuels so that absolute emission decoupling can occur in the face of growth. Other "drop-in" biofuels and their production technologies, system designs, and scale effects should continue to be investigated from a life cycle perspective – particularly for Sweden and Finland.

5.7.1 Research Implications

In Chapters 2-4, the effectiveness of the biomass sink has not been taken into explicit consideration in environmental systems analyses. Because net annual increments have been increasing and are expected to continue to increase throughout the region, it has been assumed that the replacement of fossil fuel with biofuel contributes to climate mitigation because biogenic emissions emitted at tailpipe are assumed to be immediately sequestered from sinks comprising this increment. However, the question of whether net atmospheric emissions are reduced in

actuality when the sink is removed to produce biofuel requires a more comprehensive investigation into forest dynamics and the effects of management intervention on future growth rate and age distribution. How much CO_2 might have been sequestered naturally by these sinks had they not been removed to produce biofuels? This requires full linkage of the biomass sink with biofuel emissions and a closer examination of the time dimension which is, in part, the subject of Chapter 6.

Chapter 6: The Role of the Scandinavian Boreal Forest in Climate Protection

6.1 Questions

6.1.1 Forest Dynamics and the Carbon Cycle

The practice of accounting for permanent carbon stock losses attributed to biomass production for biofuel has been commonplace in the LCA and carbon footprinting community. However, what is not practiced by biofuel researchers in these communities is detailed accounting of temporary carbon stock changes and a more complete representation of the resulting climate impact. A closer inspection of the conclusions drawn by prominent researchers of the forest carbon cycle modeling community helps shed light as to why this is the case (Schlamadinger et al., 1997; Schlamadinger & Marland, 1996a, 1996b). In the context of forest management for bioenergy, as Marland and Marland (1992) have framed it:

> *The most effective strategy for using forest land to minimize increases in atmospheric CO_2 will depend on the current status of the land, the productivity that can be expected, the efficiency with which the forest harvest is used to substitute for fossil fuels, and the time perspective of the analysis.*

As can be observed from this statement, one needs to know "the productivity [of sinks] that can *be expected*" in space and time. This consideration is outside the scope of carbon footprint analysis and LCA as it implies one must have an explicit representation of the future development of biomass sinks in time post management intervention. In other words, a full linkage of atmosphere-biosphere carbon fluxes in time and space is required. As mentioned in Chapter 2, life cycle inventory modeling is often site generic, and snapshots of historic emissions are integrated over time in impact assessment. An accurate representation of carbon cycling related to biomass growth requires a separate modeling procedure with high spatial resolution and accurate depiction of time.

However, important trade-offs between biomass harvest for displacement of fossil energy and storage in living carbon pools is not easy to quantify using existing forest carbon cycle modeling tools and LCA in isolation. LCA in its current form is not well suited to consider the complexities of forest carbon dynamics, and similarly, forest carbon cycle models do not consider auxiliary systems outside of the forest and the associated life cycle carbon inputs into these systems. Integration of LCA with forest carbon cycle modeling would improve understanding of potential contributions to climate change mitigation (McKechnie, Colombo, Chen, Mabee, & MacLean, 2011). As such, this approach is taken here to answer the following research question:

- *What is the potential of forest-biofuel to reduce GHG emissions when displacing fossil-based energy must be balanced with forest carbon implications related to biomass harvest?*

6.1.2 Forests and Climate

In addition to its role in regulating the carbon cycle, forests influence climate through exchanges of water and energy (R. Betts, 2007; Bonan, 2008; Jackson et al., 2008; Marland et al., 2003; Pielke Sr. et al., 2002).
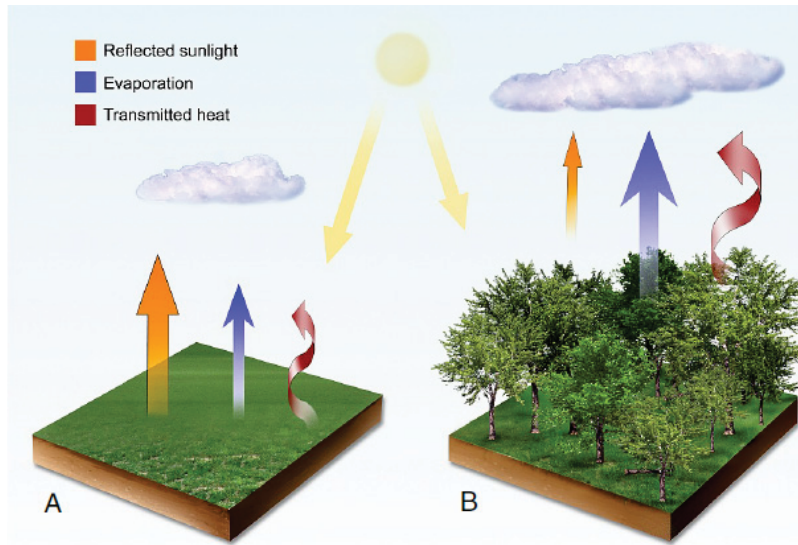
Figure 7. Generic illustration of biophysical factors influencing climate. The size of the arrow illustrates the relative magnitude of the biophysical effect across two vegetation types, grassland (A) and forests (B). Figure source: (Jackson, et al., 2008).

Biophysical factors such as reflectivity (albedo), evaporation, and surface roughness play a role in regulating surface energy fluxes and the hydrologic cycle – both affecting climate. This is exemplified for the general case of forests relative to grasslands in Figure 7, adopted from Jackson *et al.* (2008). Because grassland (Fig. 7-A) often has a higher surface albedo, it reflects more sunlight, cooling surface air temperatures relatively more than forests (Fig. 7-B). On the other hand, forests often evaporate more water and transmit more heat to the atmosphere, cooling it locally compared to grassland. More water vapor in the atmosphere can lead to a greater number and height of clouds as well as to increased convective rainfall. Further, the uneven canopy of the forest and associated higher surface roughness increases mixing and upwelling of air more so than grassland.

Land use policies for climate mitigation rarely acknowledge these biophysical factors which can alter temperature more than carbon sequestration, and sometimes in conflicting ways (Jackson, et al., 2008). The magnitude of the effects of the various biophysical factors on climate vary across regions and ecosystems (Bonan, 2008). For boreal forest regions, several studies show that many of these biophysical factors can have opposing effects on climate than that of the carbon cycle (Bala et al., 2007; R. A. Betts, 2000; Bonan & Pollard, 1992; Lyons, Jin, & Randerson, 2008; McGuire, Chapin, Walsh, & Wirth, 2006; Randerson et al., 2006; Swann, Fung, Levis, Bonan, & Doney, 2010). Many suggest that, in the case of coniferous boreal forests in high latitude regions with abundant annual snow cover, albedo is the dominant biophysical factor in opposition to the carbon cycle, particularly as it affects local radiative forcing and temperature (Bala, et al., 2007; R. Betts, 2007; R. A. Betts, 2000; McGuire, et al., 2006; Randerson, et al., 2006; Swann, et al., 2010). Thus a closer examination of the so-called "albedo effect" is warranted, particularly in the context of land managed for boreal forest biofuel production (Thompson, Adams, & Johnson, 2009), bringing us to our next research question:

- *How significant is albedo climate perturbations in relation to the carbon cycle in a scenario of increased timber harvesting for biofuel in Nordic Europe?*

6.2 Introduction to Paper V

The case of forest biofuel production and consumption in Norway is revisited so as to include the dynamics of the carbon cycle in addition to the effects of albedo changes in forests. A combined modeling approach is taken, linking LCA results from Chapter 3 with a dynamic model of prospective carbon flux changes across the full forest landscape. The landscape model is linked with albedo and climate data and fed into a radiative transfer model to approximate the magnitude of albedo climate perturbations. The mode of analysis is change oriented, the system definition is region-oriented, and the decision making scope lay at the macro-level.

Radiative Forcing Impacts of Boreal Forest Biofuels: A Scenario Study for Norway in Light of Albedo

Radiative Forcing Impacts of Boreal Forest Biofuels: A Scenario Study for Norway in Light of Albedo

Ryan M. Bright,*,[†] Anders Hammer Strømman,[†] and Glen P. Peters[‡]

[†]Industrial Ecology Program, Department of Energy and Process Engineering, Norwegian University of Science and Technology (NTNU), Trondheim, Norway

[‡]Center for International Climate and Environmental Research — Oslo (CICERO), Norway

Ⓢ *Supporting Information*

ABSTRACT: Radiative forcing impacts due to increased harvesting of boreal forests for use as transportation biofuel in Norway are quantified using simple climate models together with life cycle emission data, MODIS surface albedo data, and a dynamic land use model tracking carbon flux and clear-cut area changes within productive forests over a 100-year management period. We approximate the magnitude of radiative forcing due to albedo changes and compare it to the forcing due to changes in the carbon cycle for purposes of attributing the net result, along with changes in fossil fuel emissions, to the combined anthropogenic land use plus transport fuel system. Depending on albedo uncertainty and uncertainty about the geographic distribution of future logging activity, we report a range of results, thus only general conclusions about the magnitude of the carbon offset potential due to changes in surface albedo can be drawn. Nevertheless, our results have important implications for how forests might be managed for mitigating climate change in light of this additional biophysical criterion, and in particular, on future biofuel policies throughout the region. Future research efforts should be directed at understanding the relationships between the physical properties of managed forests and albedo, and how albedo changes in time as a result of specific management interventions.

■ INTRODUCTION

Expanding forest resource endowments combined with rising concerns for both climate change and energy security have prompted the formation of policies in Nordic Europe designed to promote the deployment of low-carbon renewable energy technologies based on boreal forest biomass, such as second generation biofuels. Assessing the climate impact of forest biofuels requires the consideration of the full range of direct and indirect climate interventions across the production life cycle, from forest management, production, and end use (combustion). Greenhouse gas-based (GHG) Life Cycle Assessment is one of the prevailing frameworks for such analyses[1−5] and has been incorporated into regional biofuel regulatory schemes[6,7] as well as national product standards.[8] However, while the framework is useful for comparing product alternatives due to its comprehensive scope, it is inherently static; that is, it provides a "snapshot" of climate impact where the snapshot is based on climate interventions that occurred over the time interval of the snapshot.[9] This means that, in the case of forest biofuel, the "cooling" impact that occurs over the forest growth period (due to the removal of CO_2 from the atmosphere) is usually not represented. In order to compensate, the "warming" impact that occurs once biofuel is combusted is often neglected because it is assumed the quantity assimilated during growth will approximately offset that which is released upon oxidation ("carbon

neutrality" principle). LCA's site- and time-generic framework limits its ability to cope with temporary carbon stock changes and issues of carbon cycling which requires full linkage of atmosphere—biosphere flows of carbon in time and space. A more accurate representation of carbon cycling related to forest growth requires a separate modeling procedure with higher spatial and temporal resolution, requisite to account for the full productivity of the forest carbon sink that can be expected when forests are managed for biofuel.[10−12] However, questions surrounding the trade-offs as to whether forests should be harvested for displacement of fossil energy or left as a sink for storing carbon in living pools are difficult to address using carbon cycle modeling tools and LCA in isolation due to their individual limitations. LCA in its current form is not well-suited to consider the complexities of forest carbon cycle, and similarly, forest carbon cycle models do not consider auxiliary systems outside of the forest and the associated life cycle carbon inputs into these systems. Integration of LCA with forest carbon cycle modeling would improve understanding of potential contributions to climate change mitigation, an approach recently embraced by McKechnie and colleagues.[13]

Further, while it is important to attribute the net carbon flux from the terrestrial sink to forest management activities, one must also be aware of the fact that—with respect to climate change mitigation objectives—forests are not limited by their ability to reduce atmospheric CO_2 concentrations. It is now becoming well-understood that boreal forests also regulate climate through a variety of biophysical mechanisms, some of which can have a significant, even dominating effect.[14−18] Biophysical factors such as reflectivity (albedo), evaporation, and surface roughness play a role in regulating surface energy fluxes and the hydrologic cycle—both affecting climate across various temporal and spatial scales.[15,19] In high latitude boreal regions with significant annual snow cover, many studies show that the albedo of forests is the dominant biophysical factor in direct opposition to the carbon cycle, and can be of the same magnitude.[20−24] It is therefore important to attribute this so-called "albedo effect"[25] to forest management and to the forest product system. Few attempts have been made to estimate the impact from a changing albedo for the primary purpose of attributing it to specific forest management projects or to specific product systems that require extensive uses of land. Schwaiger et al.[26] have estimated the net climate impact of afforestation projects in Spain when albedo is included, and Muñoz et al.[27] have sought to directly integrate albedo into an LCA framework to estimate the climate impacts of greenhouse tomatoes grown in Spain. Recently, Loarie et al.[28] and Georgescu et al.[29] have estimated the climate implications of crops grown for biofuels when albedo and other biophysical land−surface interactions are included in the modeling. For the countries of Nordic Europe that have large, well-established forestry sectors, however, no efforts have been made to include impacts from albedo changes in the climate balance of the forestry sector, neither in a dynamic perspective nor for purposes of attributing this impact to specific forest product systems like biofuels.

Objectives. In this study, we make extensive use of detailed forestry statistics combined with albedo data to construct a model that allows us to estimate the magnitude of the "albedo effect" in relation to those of the carbon cycle in a scenario of more intensive clear-cut harvesting for biofuel in Norway. Aligning with McKechnie et al.,[13] we bridge forest carbon cycle modeling with LCA results of an earlier case study of forest-biofuel in Norway[30]—which had *not* considered albedo—to account for the full spectrum of direct and indirect climate interventions in the anthropogenic production system. We address the following research questions: When considering the cumulative effects of albedo changes in forests over time, would an increase in the harvesting of forest biomass for biofuels exert a net cooling effect through increased surface albedo? How do the carbon cycle and albedo impacts associated with land use compare with the life cycle emission impacts linked to the transport fuel production and consumption system?

By building a land use model that accounts for carbon fluxes on managed productive forest areas over time, we are able to accomplish two objectives. First, by adopting an atmospheric flow perspective to biogenic carbon accounting and by way of scenario analysis we are able to quantify the extent to which forests managed for biofuel production sequester any "additional" carbon[31,32] relative to a nonbiofuel reference land use scenario. This perspective entails that biogenic carbon emissions from biofuel production and use are treated no differently than their fossil counterpart in terms of their radiative forcing impacts. By comparing changes in net biosphere carbon fluxes on the

same land areas over time, we are able to "link" emissions from bioenergy use with fluxes sequestered in living biomass.[33] Second, we now include changes in land surface albedo that accompany clear-cut harvesting and approximate the resulting net radiative forcing impacts from land use. Perturbations to the global radiation budget prior to any feedbacks, such as changes in surface albedo affecting the net shortwave radiative flux at the top of the atmosphere, can be compared directly with the effects of the carbon cycle through the concept of radiative forcing.[34] We express both albedo and GHG impacts in terms of instantaneous and time-integrated radiative forcing. Efforts are made to convey albedo forcing in terms of the more familiar emission-based metric "global warming potential (GWP)" so that albedo-forcing impacts can also be presented as a time series of discrete annual CO_2-equivalent emissions.

The overarching research objective of the study is to acquire a better understanding of the relationships between changes in the forest carbon cycle and albedo in tandem due to anthropogenic disturbance, holding all other factors constant; thus, climate change feedbacks are excluded from the analysis.

■ DATA, METHODS, AND UNCERTAINTIES

Our case-study focuses specifically on the use of Fischer−Tropsch diesel (FTD) produced in Norway. Currently, FTD is not produced or consumed in Norway. A recent report published by Norway's national Forest and Landscape Research Institute has estimated the long-term potential to increase timber extraction on productive forest areas from today's current volume of 8.2 Mm^3 to 15 Mm^3.[35] To attribute net changes in land use impacts to a future FTD system in Norway, we base scenarios of sustained future timber extraction on these figures. We assume the difference between the two figures, plus an additional 50% of the logging residues (branches, bark, foliage only), is directed entirely toward domestic FTD production for a duration of 100 years. We henceforth refer to this scenario as our *Biofuel* (*BF*) scenario. The current outtake volume and the associated land use is held constant for the same duration in order to act as a control, reference land use scenario. We refer to this scenario as our *Fossil Reference* (*FR*) scenario and assume no biofuel production, only the production of material wood products. This essentially provides the means to implement a controlled experiment for isolating the net radiative forcing impacts attributed exclusively to a change in land use for biofuels and to the transportation fuel production and consumption system.

In both scenarios, the total amount of area classified as productive forest area—or areas actively managed—remains unchanged. In other words, the increased timber extraction does not stem from bringing new forest area into production, only from the harvesting of additional forest area already classified as productive forest (i.e., no direct land use change).

Estimating Net Biogeochemical Emission Changes. *Biogenic Carbon Accounting.* Age class and species-specific area (ha), density (m^3/ha), and yield (m^3/ha/yr) data from Norway's seventh National Forest Inventory[36] are used together with biomass expansion factors[37] to develop a model that projects fluxes of carbon across five carbon pools over a 100-year management period following IPCC accounting guidelines.[38] The model is exogenously driven by defining scenarios of species-specific constant annual stemwood outtake (volume) and logging residues (see Table S3 for specific values). Yasso07 forest soil carbon model is parametrized for Norwegian conditions and used to

quantify carbon fluxes from the soil organic carbon pool.[39−41] Changes in forest management intervention (i.e., species- and age-specific harvesting and replanting) are modeled to occur at 10-year intervals, and annual changes in carbon fluxes between each intervention period are derived via linear interpolation.

According to UNFCCC reporting guidelines under the Kyoto Protocol,[42] carbon embodied in biomass outtake should be treated as an oxidized pulse emission flux in the year of extraction. To be consistent with both the atmospheric-flow approach[43] and the life cycle inventory modeling procedure in life cycle assessment (LCA)[44]—that is, accounting for flows when and where they occur within the system boundary—we model the release of carbon over time. This necessitates accounting for carbon emissions originating from the decay or combustion of biomass going forward in time. We follow this approach for biomass used as biofuel; however, for material-based wood products we find this additional complication unnecessary because in both the reference and biofuel scenarios the share of biomass harvested for material products is equal. Therefore, only carbon embodied in the share of biomass required to produce biofuel is treated as being oxidized in the year of extraction, and material wood product decay considerations are excluded from our assessment. Emissions due to natural events like forest fires are excluded due to their infrequent occurrence in Norway. Non-CO_2 biogenic GHG emissions from land use can occur via soil drainage, fertilization, and liming; however, these practices do not occur in Norwegian forestry and are also excluded from our assessment.[45]

Adopting an atmospheric-flow perspective implies that the carbon cycle climate benefit of biofuel can only come via the enhancement of terrestrial biomass sink capacity; that is, through the sequestration of an "additional carbon" flux in living biomass in an alternative land use scenario (BF) relative to some reference land use scenario (FR). By following the "Gain−Loss" method of the IPCC,[38] quantifying any "additional" input flux (henceforth labeled "net" flux because it can be either positive or negative) associated with carbon gains, ΔG, added together with the net change in output flux, ΔL, (i.e., carbon embodied in biomass outtake required for biofuel production plus net flux changes in litter and soil pools), allows us to "link" biogenic carbon emissions from the biofuel system with removals sequestered in living biomass across time and space, thereby providing the means to establish full causation of the marginal land use carbon cycle changes attributed to the combined land use plus biofuel system in Norway when going from the FR to BF scenario (i.e., $\Delta = BF − FR$).

Fossil GHG Accounting. In addition to biogenic carbon fluxes associated with land use, GHGs due to the combustion of fossil fuel occur throughout the biofuel production life cycle. By knowing the mass and heating values of the net biomass outtake fraction directed toward FTD production in the BF scenario, and by also knowing the biomass-to-FTD conversion efficiency,[30] we are able to derive a value for the quantity of FTD that is supplied over time in the scenario (TJ/yr; see Table S5 in the Supporting Information for these values). We then adapt the fossil-based life cycle GHG emission factors from ref 30 to generate a fossil emission inventory associated with this supply (CO_2, CH_4, N_2O only, also reported in Table S5). The same supply quantities together with the life cycle emission factors associated with fossil diesel production *and* use ("Well-to-Wheel", "WTW" system; also adapted from ref 30) are used to generate a reference emission inventory for the FR scenario which is needed to deduce net changes in fossil-WTW emissions, ΔWTW. The total net land use plus life cycle fossil emission changes in the BF scenario relative to the FR scenario, ΔE, can thus be expressed as:

$$\Delta E = \Delta G + \Delta L + \Delta WTW \qquad (1)$$

where the change in net annual biogeochemical emissions ΔE is equal to the sum of the net biogenic CO_2 sequestration flux change in living biomass ΔG, the net change in output flux, or the biogenic CO_2 emission embodied in the biomass required for FTD production ΔL, and the net life cycle fossil-based emission changes from transport fuel production and use, ΔWTW (i.e, avoided life cycle fossil emissions from fuel switching inclusive of emissions from fuel chains and from forestry operations). The interpretation of a positive ΔE is that increased logging plus biofuel production and use leads to a higher climate impact relative to FR. Direct biogenic carbon emissions associated with the use of FTD in vehicles (tailpipe emissions) is accounted for as part of the net land use (ΔL) flux since we assume the net biomass fraction extracted from forests is dedicated entirely toward the production of FTD in the BF scenario. We assume the remaining biogenic carbon fraction is oxidized and vented during FTD production. Non-CO_2 biogenic GHG emissions and short-lived emission components from combustion processes are excluded.

Forest Dynamics Uncertainty. An inherent uncertainty source following this approach stems from the land use scenarios. Although our model is developed using detailed national forest inventory statistics, it is not possible to predict which areas are harvested in the future. Harvest decisions in Norway lay mostly in the hands of private forest owners acting on current market conditions, and these decisions affect rotation periods, or harvest age, which in turn affects the long-term age distribution and productivity of the forest. It is also not possible to predict when future changes in routine silviculture practice might occur, such as a change in planting density or species—or from the implementation of new regimes such as thinning or fertilization for example—all of which can affect growth rates and yields. Thus several assumptions regarding future forest management operational structure are needed that affect both the carbon cycle and albedo. This includes, most notably, prescribing a 10-year harvesting regime as a function of age class and species distribution irrespective of economic criteria. Harvested areas are assumed to be immediately replanted with the same species, and the amount replanted is based on average planting densities reported during the inventory period spanning 1994−1999. As previously mentioned, our objective is to isolate and attribute climate impacts solely to the anthropogenic system, thus changes in future growth rates from the effects of climate change are not taken into consideration in the model.

We test our modeling assumptions by benchmarking the annual carbon sequestration fluxes generated by our model (see Figure S4) to those reported in a more detailed study[35] for the scenarios which are exclusive of climate change feedback considerations, finding that our model is able to project similar prognoses regarding future carbon cycle trends in forests.

Albedo Estimates. Previous studies have simulated large-scale changes from one vegetation type to another across the global landscape,[23,24,46] or have estimated changes in albedo forcing by reconstructing historical global land use change.[47,48] Pongratz et al.[49] demonstrate the importance of implementing region-specific analysis to estimate the relative magnitude of CO_2

7572

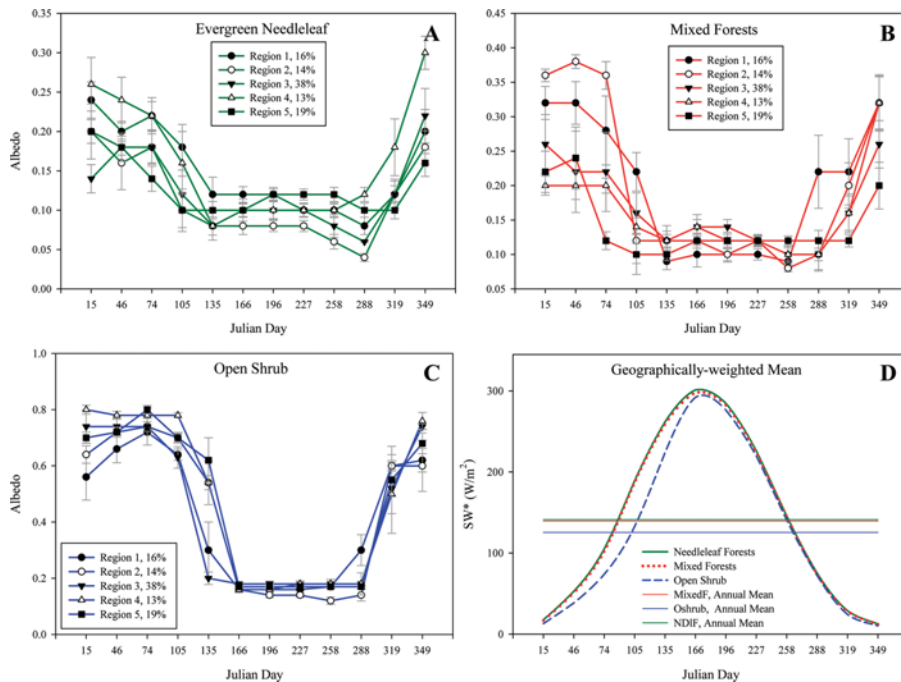dx.doi.org/10.1021/es201746b |*Environ. Sci. Technol.* 2011, 45, 7570–7580

Figure 1. (A, B, C) Nine-year monthly mean surface albedo for each land surface type for each of the five sample regions obtained from MODerate resolution Imaging Spectroradiometer (MODIS) measurements of total-sky shortwave albedo (MCD43A1 collection 5), inclusive of snow cover,[51] shown with monthly sample uncertainty at 95% (2σ) confidence. The percent logged in each region on an annual basis, both in 2009 and in the future in both logging scenarios, is shown in legends. (D) Corresponding, geographically weighted instantaneous monthly and annual average local net shortwave radiation flux at the top of the atmosphere (SW*).

and albedo change impacts of future land cover changes, particularly in boreal regions. We therefore restrict our simulation only to local regions affected by forest management activity, which requires use of a less comprehensive albedo data set. In Norway, the average clear-cut site is 2 ha, and most of the potential for increased logging is restricted to the southern half of the country[50] to five major regions (refer to Figure S1 for these regions). We make use of NASA's MODerate-resolution Imaging Spectroradiometer (MODIS) which provides monthly average total-sky shortwave surface albedo data inclusive of snow cover for an area of 500 m × 500 m (25 ha) spatial resolution.[51] The albedo data are retrieved for sites in which 100% of the pixels overlap with one of three IGBP[52] land surface types—(1) Needleleaf Forest, (5) Mixed Forest, (7) Open Shrub—as proxy land surface typology for pine- and spruce-dominant, birch-dominant, and clear-cut land surface areas, respectively. Hard-linking MODIS albedo data with specific plots is not possible due to the significant size difference between plot sizes in Norway and that afforded by MODIS, and also due to the fragmented nature of forest ownership structure and lack of publically available information regarding plot distribution as a function of individual species and age in Norway. However, satellite imaging is used in the site selection process to ensure that the sites possess canopy density characteristics representative of mature plots under management. One site representative of each surface type in each region is chosen for the purpose of capturing

geographic variances in climate affecting snow cover and phenology. Albedo data for these sites are based on the Terra and Aqua (Combined) MODIS BRDF/Albedo Model Parameter Product (MCD43A1) which are calculated using a solar zenith angle equal to the local solar noon and an optical depth of 0.2.[53] Data are retrieved for a 9-yr time series spanning January 2001 to December 2009 for each site. By adopting the monthly mean, this minimizes uncertainties related to inter- and intra-annual variability in phenology and snow cover, quality, and depth, etc., as it affects albedo. Mean values—together with climatic information on cloud cover fractions and cloud properties provided by the International Satellite Cloud Climatology Project (ISCCP)[54]— are used to derive values for the net 24-h average instantaneous short-wave (SW) radiation flux at the top of the atmosphere on the midmonth Julian day using the plane-parallel Fu-Liou Radiative Transfer Model.[55,56] The net SW flux is the difference between the downwelling and upwelling fluxes at the top of the atmosphere, henceforth denoted as SW*. Monthly SW* fluxes in each region are then weighted based on the geographic distribution of logging activity that occurred in the year 2009 to develop one geographically weighted monthly profile for each land surface type. The 9-yr mean monthly albedos for each surface type according to region, and the corresponding, geographically weighted SW* values, are shown in Figure 1. We then average this profile across the year for linking with the forest carbon model which is run in discrete annual time increments.

Table 1. Mean Annual Local SW* Values (W/m^2) Associated with the Original Sample Mean and Two Monthly Albedo Uncertainty Scenarios

	mean scenario	maximum scenario	minimum scenario
needleleaf forests	141.2	142.7	140.1
open shrub	125.4	122.0	128.7
mixed forests	139.3	140.8	137.8

Albedo Uncertainty. The 95% confidence interval shown in Figure 1 corresponds to two times the standard deviation of the monthly albedo from the 9-yr time series. We use the upper and lower uncertainty bounds for each land surface type in each region together with the climatic/cloud property data to derive both higher and lower bound monthly SW* profiles. Because we do not have spatially explicit representation of areas logged in the future, but do know the future contribution logged in each region relative to the total, monthly SW* profiles are weighted based on this share and averaged across the year for integrating with our forest carbon cycle model. This accounts for the geographic SW* variation due to differences in solar insolation and local climate/cloud cover. Upper estimates for both forest types are used together with lower bound estimates for Open Shrub (our clear-cut proxy) to create a *Maximum* uncertainty scenario—and vice versa to create a *Minimum* uncertainty scenario—to account for the full spectrum of inter and intra-annual climate variability affecting albedo, and geographic variability affecting SW*. We refer to these as our geographic uncertainty scenarios. Annual mean instantaneous local SW* values associated with these scenarios are shown in Table 1.

Radiative Forcing Model Uncertainty. The Fu-Liou radiation model has been validated previously[57,58] and used in a similar study of albedo forcing following land use change.[26] Nevertheless, we test the model by comparing results to those reported in the literature. We obtain annual mean albedo change and forcing maps for Norway from ref 59 (original study ref 47), and by picking a grid cell (see Figure S2) overlapping one of our logging regions, we benchmark the change in forcing due to a 0.001 mean albedo change with those derived using Fu-Liou, finding that our calculated results align nicely. We do the same for a grid cell in Norway reported in Figure 1 of Betts,[46] finding that a 0.2 change in mean annual albedo generates a forcing of ~12 W/m^2, aligning within the range that is reported for the same grid cell.

Climate Impact Assessment. Climate changes stemming from anthropogenic perturbations can be described as a sequence of events along a cause-effect chain beginning with biogeochemical processes, such as greenhouse gas (GHG) emissions, or biogeophysical processes, such as surface albedo change, that induce changes in Earth's radiative balance (radiative forcing) leading to changes in temperature or other climatic effects (precipitation, winds, sea level rise, etc.). The climate impact can be quantified anywhere along the cause—effect chain; however, uncertainty greatly increases as one follows forcing further down this chain.[60,61] For example, radiative forcing caused by a change in albedo can be calculated with good certainty, but how the change in energy at the surface and in the atmosphere leads to change in latent and sensible heat fluxes, local or regional pressure gradients, changes to cloudiness, air circulation, etc., is very poorly understood, and depends greatly on local geography, synoptic meteorology, etc.[15] Additionally, a biophysical land surface change can induce a weaker temperature response than a radiatively equivalent change in

CO_2 concentration due to differences in climate sensitivity, which Davin et al.[62] interpret to be a consequence of the spatial scale of a surface change forcing and the effect of nonradiative processes. We therefore limit metrics to instantaneous radiative forcing and integrated radiative forcing (iRF) from surface albedo and CO_2 flux changes using simple climate models obtained from the literature.

Net Albedo Forcing Changes in Forests over Time. Estimating albedo changes in time on managed forest areas by relying on the same parameters reported in national forest inventory statistics requires some simplified—yet important—modeling assumptions. Ideally, empirical time series data of albedo evolution as a function of species and age on areas impacted by forest management are needed for the most informed assessment, which so far does not exist for Norway. Several authors report albedo as a function of stand age following forest fire in North American boreal forests;[63−65] however, these studies focus on natural vegetation succession of the post fire environment, which is different from a managed succession where the goal is to minimize variability in tree size and condition and create spatially homogeneous, fully stocked stands of a single species.[66] At present, the question as to how forest structure and management procedures influence forest albedo remains largely unaddressed.[67] From the perspective of climate impact assessment, information on the influence of forest cover and density (as characterized by forest inventory parameters) on forest albedo is crucially needed.[68]

A recent study by Rautiainen and colleagues[68] simulated total-sky albedos with green understory and black-sky albedos with black soil for a variety of solar zenith angles of "typical" Norway spruce stands in Finland, linking forest albedo to stand structure and management practice. Nilson and Peterson[67] use airborne albedo measurements and methods of chronosequence to derive successional albedo trajectories (red, near-infrared spectra only) as best-fit lines from the measured albedos for managed boreal forests in Estonia of uniform type (>95%) and different age. Both studies report clear decreasing albedo trends in time following new plantings but results are only reported for summertime albedo.

However, Nilson and Peterson[67] and Nilson et al.[69] have noted that the two most important driving factors influencing the albedo time profile, besides species composition of the main tree story, are canopy closure (proportion of ground covered by tree crowns) and leaf area index (leaf area subtended per unit area of land; LAI). These important parameters are not reported in national forest inventories;[69] however, Nilson and Peterson[67] have simulated time series for these parameters for five even-aged, monospecies boreal forest types under management. For both parameters, the functional form of the time series was approximately linear up to a clear saturation age, at which point they then remained stable. This age varied across parameters, where the maximum canopy closure ranged from age 20 to 40 yr and maximum LAI from 20 to 70, depending on the species and site quality ("fertile" vs "infertile"; see Table S2).

Because time-series empirical data of both summer and wintertime albedo in managed, single species, even-aged stands are not available for Norway, we are forced to base our own modeling assumptions on a combination of these two driving factors to represent (i) the age at which albedo (and corresponding net radiation flux at the top of the atmosphere) returns to its original preharvest value, and (ii) the functional form of this return. Assuming albedo and SW* scale with each other, we adopt the linear time profile of canopy closure and LAI up until the age of saturation reported in ref 67 for representing the time profile of a combined summer/winter albedo decay (and thus SW*

decay since we rely on mean annual SW* values (Table 1)). Regarding the saturation age, we adapt an average of the saturation age for canopy closure and LAI for each species and site quality that is reported (Table S2).

However, we do not have albedo data and thus SW* estimates with species-specific resolution, so each species is aggregated into one of the two IGBP classifications: (1) Needleleaf Forests as a proxy for spruce/pine-dominant areas and (5) Mixed Forests for birch-dominant areas. This requires further aggregation and averaging, resulting in an albedo saturation age of 38 for Mixed and 35 for Needleleaf Forests. To test the sensitivity to changes in this parameter, we apply the higher and lower bounds of the combined mean LAI and canopy closure saturation age values, arriving at 45 and 30 for Mixed, and 40 and 30 for Needleleaf, respectively.

We assume that the annual local SW* values in Table 1 for Mixed and Needleleaf boreal forests are the maximum values associated with harvestable forests—henceforth referred to as SW*^{max}. We assume they remain constant over time after the SW* saturation age, denoted hereafter as τ. Thus the local SW* as a function of land surface type i and age a with SW*^{max} applied when a equals and exceeds τ, can be written as

$$
\text{SW*}^{local}(i, a) \\
= \begin{pmatrix} \text{SW*}^{OShrub} + \dfrac{a}{\tau}\left(\text{SW*}^{max}(i) - \text{SW*}^{OShrub}\right) & \forall a < \tau \\ \text{SW*}^{max}(i) & \forall a \geq \tau \end{pmatrix} \tag{2}
$$

where the difference in net shortwave radiation at the top of the atmosphere over clear-cut area, SW*^{OShrub}, and mature area, SW*^{max}, is adjusted linearly for ages under τ, and for any age equaling or exceeding τ, SW*^{max} of forest type i (Needleleaf or Mixed) is applied. SW*^{OShrub}, the value for Open Shrub area as a proxy for clear-cut area, is applied at age zero (at the time of harvest).

Annual *global* SW* in any annual time step t for a given scenario S may be expressed as

$$
\text{SW*}^{global,S}(t) = A_E^{-1} \sum_i \sum_a \text{SW*}^{local}(i, a) A(i, a, t) \tag{3}
$$

where the total net global shortwave radiation flux over managed areas in any time step is a summation of the age- and species-dependent energy flux in Watts (found by multiplying by the total occupied area for each species and age class, $A(i,a)$) divided by the area of the Earth's surface, A_E.

The global radiative forcing resulting from changes in surface albedo (α) when forests are managed for biofuels is the difference in the global mean annual net 24-hr instantaneous SW* flux between scenarios at each time step:

$$
\Delta \text{RF}_\alpha^{global}(t) = SW*^{global,BF}(t) - SW*^{global,FR}(t) \tag{4}
$$

Global Warming Potential (GWP). The time-integrated radiative forcing due to a single emission pulse—which is dependent on its radiative efficiency (W/m^2/kg) and its decay profile over time—is known as the absolute global warming potential (AGWP).[70] For non-CO$_2$ emissions, an exponential decay profile is used together with radiative efficiency values and atmospheric lifetimes, adopted from ref 70 (refer to SI for these values). For anthropogenic CO$_2$, a more complex decay profile based on an impulse response function (IRF) with several decay

times is used.[70] The AGWP due to a pulse GHG emission relative to that of CO$_2$ over the same integration horizon is known as its global warming potential (GWP), expressed in terms of CO$_2$-equivalents. The GWP is the predominant emission metric traditionally used in the implementation of climate policy.[61,71] Based on the Kyoto Protocol the integration is up to the time horizon of 100 years, but any time horizon may be applied.

Because of its widespread implementation, we express the contribution of radiative forcing due to albedo changes also in terms of CO$_2$-equivalent emissions. For albedo, we have already determined a global $\Delta \text{RF}_\alpha(t)$ profile due to a changing forest albedo over time (eq 4). Solving for a single pulse CO$_2$-equivalent emission at time $t = 0$ that gives the same profile would be easy; however, we need to solve for a *time series* of pulse CO$_2$-equivalent emissions that yields the identical radiative forcing time profile. This requires us to solve for $e(t)$ in the following convolution of pulse CO$_2$ emissions and a CO$_2$ decay profile, where $y(t)$ represents the CO$_2$ pulse−response function (Eq. (S1)) and k_{CO_2} is the radiative efficiency of CO$_2$ in W/m^2 per kg given a background concentration of 378 ppm:

$$
\Delta \text{RF}_\alpha^{global}(t) = \int_{t'=0}^{t'=t} k_{CO_2} e_{CO_2}(t') y_{CO_2}(t - t') \mathrm{d}t' \tag{5}
$$

We do not solve for $e(t)$ analytically but instead we solve for it discretely on an annual emission basis by rewriting eq 5 to

$$
\Delta \text{RF}_\alpha^{global}(t) = \sum_{t'=0}^{t'=t} k_{CO_2} e_{CO_2}(t') y_{CO_2}(t - t') \tag{6}
$$

Solving for the discrete annual pulse emission scenario is made easier through implementation of matrix algebra. Refer to SI for additional explanation.

To preserve transparency and reduce unnecessary complication, k_{CO_2} (radiative efficiency of CO$_2$) is held constant over the analysis time horizon. We test the potential bias that this constant factor introduces when applying the logarithmic expression reported in Myhre et al.[72] and find that it does not affect our conclusions, as the concentration increases associated with our CO$_2$ emission scenarios are negligible.

Scenario Emissions and Integrated Radiative Forcing (iRF). We want to know the time-integrated radiative forcing (iRF) after 100 years expressed in W/m^2 due to annual pulse emissions occurring each year over the 100-year emission scenario. For any single kg emission of substance x of our emission scenario (Table S5), we know that AGWP at time t is

$$
\text{AGWP}_x(t) = \int_0^t \Delta \text{RF}_x(t) \mathrm{d}t \tag{7}
$$

We can find the iRF of an emission scenario S at time t for substance x through convolution of $e_x(t)$ with AGWP$_x(t)$:

$$
\text{iRF}_x^S(t) = \int_{t'=0}^{t'=t} e_x(t') \text{AGWP}_x(t - t') \mathrm{d}t'
$$

$$
= \int_{t'=0}^{t'=t} k_x e_x(t') y(t - t') \mathrm{d}t' \tag{8}
$$

RESULTS

For the first ∼30 years, negative radiative forcing due to cumulative albedo changes in forests for the *Mean* geographic uncertainty scenario (dashed green line, Figure 2A) offsets
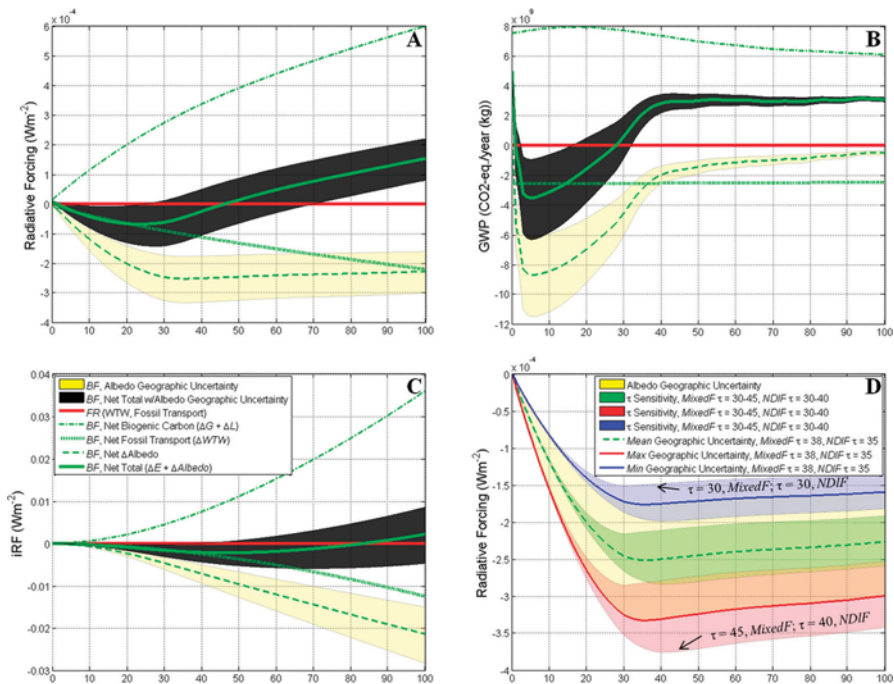
Figure 2. Changes in (A) radiative forcing, (B) global warming potential (GWP), and (C) integrated radiative forcing (iRF) associated with the forest Biofuel (*BF*) scenario. "Geographic albedo uncertainty," shown in yellow, represents all possible results affiliated with SW*max values that are between those presented in Table 1 for the Minimum and Maximum scenario for each land surface type. (D) Radiative forcing from albedo change is sensitive to albedo saturation ages, tau (τ), associated with Needleleaf (NDlF) and Mixed Forests (MixedF).

positive forcing from increased biogenic CO_2 emissions (dashed-dotted green line, Figure 2A). The albedo-forcing profile eventually stabilizes at around the time corresponding to τ, at which point the forcing from avoided fossil fuel emission (ΔWTW, dotted green line, Figure 2A) begins to play the more active offset role. The net combined effect results in a forcing benefit for 45 years for the *Mean* scenario (solid green line, Figure 2A), with the benefit brought back or pushed forward in time to 25—70 years (black band) depending on the *Minimum* and *Maximum* geographic uncertainty scenario and for all combinations of mean annual SW* values between those presented in Table 1 for each land surface type.

The effect of the stabilizing net albedo forcing profile, when translated into annual CO_2-equivlent emission pulses, is illustrated in Figure 2B (dashed green line). Due to the long lifetime of CO_2 in the atmosphere following an initial pulse, and the forcing it induces over 100 years ($\sim$36% of the initial CO_2 in the first year remains in the atmosphere at year 100), matching the $\Delta RF_\alpha^{global}(t)$ profile with annual CO_2—equiv pulses equates to rapidly diminishing emissions after the first decade.

In terms of time-integrated radiative forcing (iRF) we see that the combined effect of a changing forest albedo plus fossil fuel substitution leads to a near-climate-neutral system over the full analysis time horizon. The crossover point, or the point in time at which the Biofuel scenario becomes worse relative to the *Fossil Reference* scenario (red line, Figure 2C) is around 40 years when the SW* parameters of the *Minimum* scenario are applied

(top of the black band) and 85 years for the *Mean* scenario (solid green, Figure 2C). For the *Maximum* scenario, a climate benefit over the entire period is realized (bottom of the black band).

The effect when τ is adjusted can be seen in Figure 2D. Adjustments to this parameter, or the age at which the forest SW* values return to their original (SW*max(i), Table 1) following clear-cut harvesting and replanting, can have a substantial effect on the albedo forcing time profile affecting both the magnitude and duration of the carbon cycle offset benefits. When combining this uncertainty with the geographic uncertainty, radiative forcing benefits can vary by as much as a factor of 2.5. For example, the mean annual SW*max parameters of the *Minimum* scenario combined with decreases to τ (top of blue band, Figure 2D) leads to a much higher forcing profile than when the *Maximum* scenario's parameters combined with an increase in τ are applied (bottom of red band, Figure 2D).

■ DISCUSSION

Important general conclusions can be drawn from our results which may have significant consequences for how forests might be managed for biofuels in light of the "albedo effect".[25] We showed that the negative albedo forcing due to cumulative effects of albedo changes in forests equaled or exceeded the positive carbon cycle forcing in the short term up until around the age of τ, implying that, by shortening rotation periods, it may be possible to realize desirable forcing benefits from albedo changes

that extend further in time. We also showed that, after τ, the time around which the forest reaches a new steady-state with respect to albedo forcing, the fossil substitution effect (ΔWTW) begins to play the more active carbon offset role. This implies that it may be equally as important to enhance the productive capacity of the forest carbon sink simultaneously in the short term via changes in other management practices, such as, for example, though fertilization, species switching, or through intensification of regeneration efforts—particularly when the forest product substitutes the fossil reference product at low efficiency, as is the case for biofuel. However, carbon management decision variables affecting short-term productivity like species switching or regeneration efforts, for example, could conflict with desirable albedo benefits.[73] Carbon cycle and albedo trade-offs as a function of specific management intervention ought to be investigated in greater detail in future work.

In this regard, more research is required for the most informed decision making regarding boreal forest management in Norway, particularly given the importance of other biophysical factors.[14,17,18] Currently in Norway a large amount of attention regarding forest management and climate is limited to the carbon cycle—both in research[74] and in recent policy.[75] How management decisions may impact other biophysical factors influencing climate by way of the surface energy budget and the hydrologic cycle ought also to be investigated. Forests, in general, often evaporate more water and transmit more heat to the atmosphere, cooling it locally compared to open shrub or cropland.[17] More water vapor in the atmosphere can lead to a greater number and height of clouds which can affect both the long-wave and short-wave radiative balance.[29,76] Swann et al.[76] for example, estimate that the radiative energy imbalance due to the radiative forcing effects of water vapor can be of the same sign and order of magnitude as short-wave forcing from albedo changes in deciduous boreal forests. Many of these biophysical factors can affect local and global climate in different ways[77,78] and may serve to either offset or reinforce carbon cycle benefits.[14,15] Deforestation of boreal forests in high latitude regions, in addition to causing local cooling, may also produce cooling elsewhere remotely through circulation.[78]

Global climate models with an interactive terrestrial biosphere are needed to fully understand these dynamic and often non-linearly related climate effects, many of which, particularly nonradiative effects such as those which act directly via surface moisture fluxes, cannot be compared directly with the effects of the carbon cycle.[15,34] Currently no metric for quantifying non-radiative climate forcings has been accepted,[79] and more complex models may also be required to understand how carbon cycle and albedo dynamics might change under a changing climate when the resulting feedbacks are included.

Nevertheless, we showed how it is possible to estimate the net radiative forcing balance attributed to specific product systems such as boreal forest biofuels in Norway in a broad, integrated modeling perspective when albedo and time considerations are included in impact assessment. Our efforts have led to a better understanding of the long-term relationships between the carbon cycle and albedo as they are affected exclusively by anthropogenic disturbance, *ceteris paribus*. Results reinforce general conclusions drawn in other literature[21–24,46,64] that the cooling effects of albedo change in high latitude boreal regions such as Norway are important to consider before sound land use and bioenergy policies are to be implemented in those regions. However, given the uncertainties in the geographic distribution of logging activity

and local climate variables affecting albedo and SW* fluxes, the magnitude of the annual forcing due to albedo changes in forests was shown to vary (Figure 2A). Further, results were found to be highly sensitive to albedo dynamics following clear-cut harvesting, in particular the time evolution of albedo decay (Figure 2D). This points to a clear scientific need for developing an empirical understanding of how management practice influences physical properties of the forest canopy and its albedo, and how albedo changes in time as a result of specific management interventions such as thinning or delayed regeneration, for example. Alternatively, radiative transfer modeling can be applied as an efficient tool for predicting the influence of various management practices on vegetation cover albedo.[67–69] The concept of reporting "reflectance tables"[67] similar to yield tables in forest inventory statistics would facilitate robust analyses regarding optimal forest management strategies for achieving climate change mitigation objectives. Broader assessments that seek to advance climate modeling of forest product systems such as biofuels require moving beyond stand alone LCA/carbon footprint type assessments toward integrated frameworks that are not restricted to emission-based metrics and that have higher temporal and spatial resolution.

■ ASSOCIATED CONTENT

ⓢ Supporting Information. Additional information regarding albedo and climate modeling methodology, life cycle emission data, additional results tables and figures, and important parameter values. This material is available free of charge via the Internet at http://pubs.acs.org.

■ AUTHOR INFORMATION

Corresponding Author
*E-mail: ryan.m.bright@ntnu.no; mail: Høgskoleringen 5, Realfagbygget E1, 7491 Trondheim, Norway; phone: +47 73 59 89 72; fax: +47 73 59 89 43.

■ ACKNOWLEDGMENT

■ REFERENCES

(1) Kammen, D. M.; Farrell, A. E.; Plevin, R. J.; Jones, A. D., Energy and greenhouse impacts of biofuels: A framework for analysis. In *Biofuels: Linking Support to Performance*; OECD/ITF: Paris, 2008; pp 41–74.

(2) Cruetzig, F. S.; Kammen, D. M., Chapter 26: Getting the carbon out of transportation fuels. In *Global Sustainability: A Nobel Cause*; Schellnhuber, H.-J., Molina, M., Stern, N., Huber, V., Kadner, S., Eds.; Cambridge University Press: Cambridge, 2010; pp 307–318.

(3) Cherubini, F.; Strømman, A. H. Life cycle assessment of bioenergy systems: State of the art and future challenges. *Bioresour. Technol.* **2010**, *102* (2), 437–451.

(4) van der Voet, E.; Lifset, R. J.; Luo, L. Life-cycle assessment of biofuels, convergence and divergence. *Biofuels* **2010**, *1* (3), 435–449.

7577

dx.doi.org/10.1021/es201746b |*Environ. Sci. Technol.* 2011, 45, 7570–7580

(5) Bringezu, S.; Schütz, H.; O'Brien, M.; Kauppi, L.; Howarth, R. W.; NcNeely, J. *Towards Sustainable Production and Use of Resources: Assessing Biofuels*; United Nations Environment Programme (UNEP), International Panel for Sustainable Resource Management (IPSRM): Paris, July, 2009.

(6) State of California. *Executive Order S-01-07: Low Carbon Fuel Standard*; Sacramento, CA, 2007.

(7) European Commission. *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 and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC*; Brussels, 2009.

(8) British Standards Institute. *PAS 2050:2008 Specification for the assessment of the life cycle greenhouse gas emissions of goods and services*; British Standards Institute: London, UK, October, 2008.

(9) Udo de Haes, H. A.; Jolliet, O.; Finnveden, G.; Hauschild, M. Z.; Krewitt, W.; Müller-Wenk, R. Best available practice regarding impact categories and category indicators in life cycle impact assessment: Background document for the second working group on life cycle impact assessment of SETAC-Europe (WAI-2). *Int. J. Life Cycle Assess.* **1999**, *4* (2), 66–74.

(10) Schlamadinger, B.; Marland, G. The Role of Forest and Bioenergy Strategies in the Global Carbon Cycle. *Biomass Bioenergy* **1996**, *10* (5/6), 275–300.

(11) Schlamadinger, B.; Marland, G. Full fuel cycle carbon balances of bioenergy and forestry options. *Energy Convers. Manage.* **1996**, *37* (6−8), 813–818.

(12) Schlamadinger, B.; Apps, M.; Bohlin, F.; Gustavsson, L.; Jungmeier, G.; Marland, G.; Pingoud, K.; Savolainen, I. Towards a standard methodology for greenhouse gas balances of bioenergy systems in comparison with fossil energy systems. *Biomass Bioenergy* **1997**, *13* (6), 359–375.

(13) McKechnie, J.; Colombo, S.; Chen, J.; Mabee, W. E.; MacLean, H. L. Forest Bioenergy or Forest Carbon? Assessing Trade-Offs in Greenhouse Gas Mitigation with Wood-Based Fuels. *Environ. Sci. Technol.* **2011**, *45*, 789–795.

(14) Betts, R. Implications of land ecosystem-atmosphere interactions for strategies for climate change adaptation and mitigation. *Tellus B* **2007**, *59* (3), 602–615.

(15) Pielke, R. A., Sr.; Marland, G.; Betts, R. A.; Chase, T. N.; Eastman, J. L.; Niles, J. O.; Niyogi, D. S.; Running, S. W. The influence of land-use change and landscape dynamics on the climate system: Relevance to climate-change policy beyond the radiative effect of greenhouse gases. *Phil. Trans. R. Soc. London A* **2002**, *360*, 1705–1719.

(16) Pielke, R. A., Sr.; Avissar, R.; Raupach, M. R.; Dolman, A. J.; Zeng, X.; Denning, A. S. Interactions between the atmosphere and terrestrial ecosystems: Influence on weather and climate. *Global Change Biol.* **1998**, *4*, 461–475.

(17) Jackson, R. B.; Randerson, J. T.; Canadell, J. G.; Anderson, R. G.; Avissar, R.; Baldocchi, D. D.; Bonan, G. B.; Caldeira, K.; Diffenbaufh, N. S.; Field, C. B.; Hungate, B. A.; Jobbágy, E. G.; Kueppers, L. M.; Nosetto, M. D.; Pataki, D., E., Protecting climate with forests. *Environ. Res. Lett.* **2008**, *3*, 044006.

(18) Bonan, G. B. Forests and Climate Change: Forcings, Feedbacks, and the Climate Benefits of Forests. *Science* **2008**, *320*, 1444–1449.

(19) Marland, G.; Pielke, R. A.; Apps, M.; Avissar, R.; Betts, R. A.; Davis, K. J.; Frumhoff, P. C.; Jackson, S. T.; Joyce, L. A.; Kauppi, P.; Katzenberger, J.; MacDicken, K. G.; Neilson, R. P.; Niles, J. O.; Niyogi, D. S.; Norby, R. J.; Pena, N.; Sampson, N.; Xue, Y. The climatic impacts of land surface change and carbon management, and the implications for climate-change mitigation policy. *Clim. Policy* **2003**, *3*, 149–157.

(20) McGuire, A. D.; Chapin, F. S.; Walsh, J. E.; Wirth, C. Integrated Regional Changes in Arctic Climate Feedbacks: Implications for the Global Climate System. *Annu. Rev. Environ. Resour.* **2006**, *31* (1), 61–91.

(21) Bonan, G. B.; Pollard, D. Effects of boreal forest vegetation on global climate. *Nature* **1992**, *359* (6397), 716–718.

(22) Claussen, M.; Brovkin, V.; Ganopolski, A. Biogeophysical versus biogeochemical feedbacks of large-scale land cover change. *Geophys. Res. Lett.* **2001**, *28* (6), 1011–1014.

(23) Bala, G.; Caldeira, K.; Wickett, M.; Phillips, T. J.; Lobell, D. B.; Delire, C.; Mirin, A. Combined climate and carbon-cycle effects of large-scale deforestation. *Proc. Natl. Acad. Sci. U.S.A.* **2007**, *104* (16), 6550–6555.

(24) Bathiany, S.; Claussen, M.; Brovkin, V.; Raddatz, T.; Gayler, V. Combined biogeophysical and biogeochemical effects of large-scale forest cover changes in the MPI earth system model. *Biogeosciences* **2010**, *7*, 1383–1399.

(25) Thompson, M.; Adams, D.; Johnson, K. N. The Albedo Effect and Forest Carbon Offset Design. *J. Forest.* **2009**, *107* (8), 425–431.

(26) Schwaiger, H. P.; Bird, D. N. Integration of albedo effects caused by land use change into the climate balance: Should we still account in greenhouse gas units? *Forest Ecol. Manage.* **2010**, *260* (3), 278–286.

(27) Muñoz, I.; Campra, P.; Fernández-Alba, A. R. Including CO_2-emission equivalence of changes in land surface albedo in life cycle assessment. Methodology and case study on greenhouse agriculture. *Int. J. Life Cycle Assess.* **2010**, *15*, 672–681.

(28) Loarie, S. R.; Lobell, D. B.; Asner, G. P.; Mu, Q.; Field, C. B. Direct impacts on local climate of sugar-cane expansion in Brazil. *Nature Clim. Change* **2011**, *1*, 105–109, DOI: 10.1038/nclimate1067.

(29) Georgescu, M.; Lobell, D. B.; Field, C. B. Direct climate effects of perennial bioenergy crops in the United States. *Proc. Natl. Acad. Sci. U.S.A.* **2011**, *108* (11), 4307–4312.

(30) Bright, R. M.; Strømman, A. H.; Hawkins, T. R. Environmental Assessment of Wood-based Biofuel Production and Consumption Scenarios in Norway. *J. Ind. Ecol.* **2010**, *14* (3), 422–439.

(31) Searchinger, T. D.; Hamburg, S. P.; Melillo, J.; Chameides, W.; Havlik, P.; Kammen, D. M.; Likens, G. E.; Lubowski, R. N.; Obersteiner, M.; Oppenheimer, M.; Philip Robertson, G.; Schlesinger, W. H.; David Tilman, G. Fixing a Critical Climate Accounting Error. *Science* **2009**, *326* (5952), 527–528.

(32) Searchinger, T. D. Biofuels and the need for additional carbon. *Environ. Res. Lett.* **2010**, *5* (2), 10.

(33) Marland, G. Accounting for Carbon Dioxide Emissions from Bioenergy Systems. *J. Ind. Ecol.* **2010**, *14* (6), 866–869.

(34) Betts, R.; Falloon, P. D.; Goldewijk, K. K.; Ramankutty, N. Biogeophysical effects of land use on climate: Model simulations of radiative forcing and large-scale temperature change. *Agric. Forest Meteorol.* **2007**, *142*, 216–233.

(35) Astrup, R.; Dalsgaard, L.; Eriksen, R.; Hylen, G. *Utviklingsscenarioer for karbonbinding i norges skoger [Development scenarios for carbon binding in Norwegian forests]*; Norwegian Forest and Landscape Institute: Ås, 2010.

(36) European Forest Institute. National Forest Inventory (NFI), Norway, 1994(95)-1999. In *The EFISCEN Inventory Database*; The European Forest Institute (EFI); Accessed May 19, 2010; http://www.efi.int/portal/virtual_library/databases/efiscen/inventory_database/#searchdb.

(37) Lehtonen, A.; Mäkipää, R.; Heikkinen, J.; Sievänen, R.; Liski, J. Biomass expansion factors (BEFs) for Scots pine, Norway spruce, and birch according to stand age for boreal forests. *Forest Ecol. Manage.* **2004**, *188*, 211–224.

(38) Aalde, H.; Gonzalez, P.; Gytarsky, M.; Krug, T.; Kurz, W. A.; Lasco, R. D.; Martino, D. L.; McConkey, B. G.; Ogle, S.; Paustian, K.; Raison, J.; Ravindranath, N. H.; Schoene, D.; Smith, P.; Somogyi, Z.; van Amstel, A.; Verchot, L. Chapter 2: Generic Methodologies Applicable to Multiple Land-use Categories. In *2006 IPCC Guidelines for National Greenhouse Gas Inventories, Vol. 4: Agriculture, Forestry, and Other Land Use*; Eggleston, S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K., Eds.; The Intergovernmental Panel on Climate Change (IPCC): Japan, 2006.

(39) Liski, J.; Tuomi, M.; Palosuo, T. *Yasso07 -- soil carbon model*; Finnish Environment Institute: Helsinki, 2009.

(40) Liski, J.; Palosuo, T.; Peltoniemi, M.; Sievänen, R. Carbon and decomposition model Yasso for forest soils. *Ecol. Model.* **2005**, *189*, 168–182.

7578

dx.doi.org/10.1021/es201746b |*Environ. Sci. Technol.* 2011, 45, 7570–7580

(41) Tuomi, M.; Thum, T.; Järvinen, H.; Fronzek, S.; Berg, B.; Harmon, M. E.; Trofymow, J. A.; Sevanto, S.; Liski, J. Leaf litter decomposition -- Estimates of global variability based on Yasso07 model. *Ecol. Model.* **2009**, *220* (23), 3362–3371.

(42) UNFCCC. *The Kyoto Protocol to the United Nations Framework Convention on Climate Change*; UNEP/WMO, 1997.

(43) Brown, S.; Lim, B.; Schlamadinger, B. *Evaluating Approaches for Net Emissions of Carbon Dioxide from Forest Harvesting and Wood Products*; IPCC: Dakar, Senegal, May 1998.

(44) ISO. Environmental Management-Lifecycle Assessment-Principles and Framework. In *ISO 14044*, 2nd ed.; International Organization for Standardization, 2006; Vol. 14044.

(45) Rypdal, K.; Bloch, V. V. H.; Flugsrud, K.; Gobakken, T.; Hoem, B.; Tomter, S. M.; Aalde, H. *Emissions and removals of greenhouse gases from land use, land use change, and forestry in Norway*; Center for International Climate and Environmental Research -- Oslo; Norwegian Institute of Land Inventory; Statistics Norway: Ås/Oslo, 2005.

(46) Betts, R. A. Offset of the potential carbon sink from boreal forestation by decreases in surface albedo. *Nature* **2000**, *408*, 187–190.

(47) Myhre, G.; Kvalevåg, M. M.; Schaaf, C. B. Radiative forcing due to anthropogenic vegetation change based on MODIS surface albedo data. *Geophys. Res. Lett.* **2005**, *32*, L21410.

(48) Pongratz, J.; Raddatz, T.; Reick, C. H.; Esch, M.; Claussen, M. Radiative forcing from anthropogenic land cover change since A.D. 800. *Geophys. Res. Lett.* **2009**, *36*, L02709.

(49) Pongratz, J.; Reick, C. H.; Raddatz, T.; Caldeira, K.; Claussen, M. Past land use decisions have increased mitigation potential of reforestation. *Geophys. Res. Lett.* **2011**, *38*, L15701, 5 pp.

(50) Vennesland, B.; Hobbelstad, K.; Bolkesjø, T.; Baardsen, S.; Lileng, J.; Rolstad, J. *Forest Resources in Norway 2006: Possibilities and actual strategies for increased harvesting [In Norwegian]*; Norwegian Forest and Landscape Institute: Ås, 2006; Accessed November 17, 2008; http://www.skogoglandskap.no/filearchive/5.pdf.

(51) Oak Ridge National Laboratory, Distributed Active Archive Center (ORNL DAAC). MODIS subsetted land products, Collection 5; ORNL DAAC: Oak Ridge, TN, 2010; Available at http://daac.ornl.gov/MODIS/modis.html]; Accessed 30 September, 2010.

(52) IGBP is the International Geosphere-Biosphere Programme providing a standard framework for the systematic classification of land surface typology according to shared physical, chemical, and biological features. The MODIS BRDF/Albedo product (MCD43A1) is integrated with MODIS Land Cover Product (MOD12, Collection 5) which is based on IGBP Classification Type_1.

(53) BRDF is the "Bidirectional Reflectance Distribution Function" giving the reflectance of a target as a function of illumination geometry and viewing property. The BRDF depends on wavelength, and is determined by the structural and optical properties of the surface such as shadow-casting, multiple scattering, mutual shadowing, transmission, reflection, absorption and emission by surface elements, facet orientation distribution, and facet density. For more information, see http://www-modis.bu.edu/brdf/userguide/index.html.

(54) ISCCP International Satellite Cloud Climatology Project. http://isccp.giss.nasa.gov/.

(55) Fu-Liou. Fu-Liou online 200507 (Diurnal Simulation); 2005; http://snowdog.larc.nasa.gov/cgi-bin/rose/flp200503/sun/flsun.cgi.

(56) Fu, Q.; Liou, K. N. Parameterization of the Radiative Properties of Cirrus Clouds. *J. Atmos. Sci.* **1993**, *50* (13), 2008–2025.

(57) Rose, F. G.; Charlock, T. P. *New Fu-Liou Code Tested with ARM Raman Lidar and CERES in pre-CALIPSO Exercise*. Extended abstract for the 11th Conference on Atmospheric Radiation (AMS), Ogden, UT, USA, 3—7 June, 2002.

(58) Huang, J.; Fu, Q.; Su, J.; Tang, Q.; Minnis, P.; Hu, Y.; Yi, Y.; Zhao, Q. Taklimakan dust aerosol radiative heating derived from CALIPSO observations using the Fu-Liou radiation model with CERES constraints. *Atmos. Chem. Phys.* **2009**, *9*, 4011–4021.

(59) Kvalevåg, M. M. Changes in Surface Albedo and Radiative Forcing in Norway. Pesonal communication, November 10, 2010.

(60) Shine, K. P.; Fuglestvedt, J. S.; Hailemariam, K.; Stuber, N. Alternatives to the global warming potential for comparing climate impacts of emissions of greenhouse gases. *Climatic Change* **2005**, *68*, 281–302.

(61) Fuglestvedt, J. S.; Shine, K. P.; Berntsen, T.; Cook, J.; Lee, D. S.; Stenke, A.; Skeie, R. B.; Velders, G. J. M.; Waitz, I. A. Transport impacts on atmosphere and climate: Metrics. *Atmos. Environ.* **2009**, *44* (37), 4648–4677.

(62) Davin, E. L.; de Noblet-Ducoudré, N.; Friedlingstein, P. Impact of land cover change on surface climate: Relevance of the radiative forcing concept. *Geophys. Res. Lett.* **2007**, *34*, L13702.

(63) Amiro, B. D.; Orchansky, A. L.; Barr, A. G.; Black, T. A.; Chambers, S. D.; Chapin, F. S., III; Goulden, M. L.; Litvak, M.; Liu, H. P.; McCaughey, J. H.; McMillan, A.; Randerson, J. T. The effect of post-fire stand age on the boreal forest energy blance. *Agric. Forest Meteorol.* **2006**, *140*, 41–50.

(64) Randerson, J. T.; Liu, H.; Flanner, M. G.; Chambers, S. D.; Jin, Y.; Hess, P. G.; Pfister, G.; Mack, M. C.; Treseder, K. K.; Welp, L. R.; Chapin, F. S.; Harden, J. W.; Goulden, M. L.; Lyons, E.; Neff, J. C.; Schuur, E. A. G.; Zender, C. S. The Impact of Boreal Forest Fire on Climate Warming. *Science* **2006**, *314*, 1130–1132.

(65) Lyons, E.; Jin, Y.; Randerson, J. T. Changes in surface albedo after fire in boreal forest ecosystems of interior Alaska assessed using MODIS satellite observations. *J. Geophys. Res.* **2008**, *13*, G02012.

(66) Franklin, J. F.; Spies, T. A.; Van Pelt, R.; Carey, A. B.; Thornburgh, D. A.; Rae Berg, D.; Lindenmayer, D. B.; Harmon, M. E.; Keeton, W. S.; Shaw, D. C.; Bible, K.; Chen, J. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. *Forest Ecol. Manage.* **2002**, *155*, 399–423.

(67) Nilson, T.; Peterson, U. Age Dependence of Forest Reflectance: Analysis of Main Driving Factors. *Remote Sens. Environ.* **1994**, *48*, 319–331.

(68) Rautiainen, M.; Stenberg, P.; Mõttus, M.; Manninen, T. Radiative transfer simulations link boreal forest structure and shortwave albedo. *Boreal Environ. Res.* **2011**, *16*, 91–100.

(69) Nilson, T.; Kuusk, A.; Lang, M.; Lükk, T. Forest Reflectance Modeling: Theoretical Aspects and Applications. *Ambio* **2003**, *32* (8), 535–541.

(70) Forster, P.; Ramaswamy, V.; Artaxo, P.; Berntsen, T.; Betts, R.; Fahey, D. W.; Haywood, J.; Lean, J.; Lowe, D. C.; Myhre, G.; Nganga, J.; Prinn, R.; Raga, G.; Schulz, M.; Van Dorland, R. Changes in atmospheric consituents and in radiative forcing. In *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*; Soloman, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B., Tignor, M., Miller, H. L., Eds.; Cambridge University Press: Cambridge, U. K. and New York, 2007.

(71) IPCC. *Summary Report of the IPCC Expert Meeting on the Science of Alternative Metrics*; IPCC, Oslo, Norway, March 18—20, 2009; Available at www.ipcc.ch/meetings/session30/doc13.pdf.

(72) Myhre, G.; Highwood, E. J.; Shine, K. P.; Stordal, F. New estimates of radiative forcing due to well mixed greenhouse gases. *Geophys. Res. Lett.* **1998**, *25* (14), 2715–2718.

(73) Thompson, M. P.; Adams, D.; Sessions, J. Radiative forcing and the optimal rotation age. *Ecol. Econ.* **2009**, *68*, 2713–2720.

(74) Holtsmark, B. *Use of wood fuels from boreal forests will create a biofuel carbon debt with a long payback time*; Statistics Norway, Research Department, November, 2010.

(75) Norwegian Ministry of Land Use and Agriculture. *Climate challenges -- land use a part of the solution*; St. meld. nr. 39; Ministry of Land Use and Agriculture: Oslo, 2009; p 177.

(76) Swann, A. L.; Fung, I. Y.; Levis, S.; Bonan, G. B.; Doney, S. C. Changes in Arctic vegetation amplify high-latitude warming through the greenhouse effect. *Proc. Natl. Acad. Sci. U.S.A.* **2010**, *107* (4), 1295–1300.

(77) Gedney, N.; Valdes, P. J. The effect of Amazonian deforestation on the northern hemisphere circulation and climate. *Geophys. Res. Lett.* **2000**, *27* (19), 3053–3056.

7579

dx.doi.org/10.1021/es201746b |*Environ. Sci. Technol.* 2011, 45, 7570–7580

(78) Chase, T. N.; Pielke, R. A., Sr.; Kittel, T. G. F.; Nemani, R. R.; Running, S. W. Simulated impacts of historical land cover changes on global climate in northern winter. *Clim. Dynam.* **2000**, *16*, 93–105.

(79) Pielke, R. A., Sr.; Adegoke, J.; Beltrán-Przekurat, A.; Hiemstra, C. A.; Lin, J.; Nair, U. S.; Niyogi, D.; Nobis, T. E. An overview of regional land-use and land-cover impacts on rainfall. *Tellus B* **2007**, *59* (3), 587–601.

6.4 Uncertainties and Limitations

Albedo modeling presents the largest source of uncertainty, particularly the albedo time profile post harvest. This is owed to an underrerporting of empirical albedo measurement data in managed forests of uniform type and species in Nordic boreal regions. The study is also limited by the exclusion of other biophysical factors and non-linear climate forcings.

6.5 Summary and Conclusions

The research performed in this chapter and conclusions drawn therein regarding the climate impacts of forest biofuels are in opposition to conclusions drawn from previous chapters. Full linkage of carbon sources with sinks and an explicit representation of time in climate impact assessment showed that forest biofuels, for the Norwegian case, are likely to play a limited, even counterproductive role in contributing to climate change mitigation policy, particularly over the medium- and longer-terms. Albedo played a cooling offset role in the short-term; however, at a constant, sustained harvest level, over the long term the forest reached a new steady-state with respect to albedo forcing, and the carbon cycle warming impacts from lost sinks and increased biogenic carbon dioxide emissions dominated the net radiative forcing profile.

6.5.1 Research Implications

These conclusions have important implications for forest management and biofuel policy. Across the landscape it will be difficult to manage forests in ways that maximize short-term albedo benefits while simultaneously enhancing the long-term productivity of the biomass sink such that increased timber extraction for the production of biofuels leads to sustained long-term climate benefits. Identifying optimal forest management strategies that maximize both short-term albedo benefits and long-term carbon cycle benefits – such as a harvesting regime that prioritizes low density areas (m^3/ha) and a silviculture regime that enhances site productivity on those same areas (m^3/ha/yr) – ought to be the subject of future research. Identifying these strategies will undoubtedly require the application of broader modeling frameworks that have high temporal and spatial resolution of the most dominant physical parameters and management decision variables affecting surface albedo and the carbon cycle, both at the stand and landscape levels.

The work performed in this chapter may be viewed as an important environmental systems analysis research contribution as it demonstrates the importance of applying broader analytical frameworks to answer questions about the climate mitigation efficacy of specific forest based products like transportation biofuels. The literature review of this work also points to the existence of a large gap in the literature surrounding albedo modeling of managed boreal forests, particularly the characterization of albedo as a function species composition and stand age resulting from human disturbance.

Chapter 7: Summary and Outlook

Summary

Peak oil combined with sustained greenhouse gas emission growth call for fundamental shifts in Nordic road transportation technologies and policy. This thesis has provided new insights into how forest biofuels would play a role in mitigating climate change and fossil fuel dependency by analyzing, interpreting, simulating, and communicating the problem from multiple angles combining environmental, economic, technological, and policy perspectives.

In Chapters 2 and 3, exemplified by the Norwegian cases, it was found that the production and use of forest biofuels pose no significant additional risks to human health and other environmental impacts like eutrophication and acidification relative to conventional fuels based on oil. Impacts in these categories varied across product systems and data sets, but in all cases were shown to fall within the same order of magnitude as those from current fossil systems. It was found that conversion processes based on thermochemical production platforms have lower life cycle emissions, and, for all systems evaluated, both up- and downstream transportation processes contributed to the majority of the fuel production system impacts across all impact categories considered.

Life cycle greenhouse gas emissions due to the production of forest biofuels were found to be slightly higher than conventional fossil transport fuels in most cases, implying that emission benefits stem from the use phase and the recycling of carbon by terrestrial biomass sinks when stocks are not depleted. The largest source of greenhouse gas emission in the production system stemmed from wood chip storage processes, but this impact is a modeling artifact stemming from a choice in system design and can be mitigated.

In Chapter 3, risk of carbon leakage was identified when greenhouse gas impacts of Norwegian forest biofuels were quantified following a consumption-based approach which captured the emissions occurring outside Norway in a scenario where Norwegian pulp and paper production ceased – but demand for pulp and paper products remained. While this scenario was not due to the diversion of resources from the existing wood industry as a result of increased biofuel production, one could easily envision such a scenario in which a competing biofuels industry does cause global production shifts, reinforcing the need to implement biofuel policy based on careful assessments of local resource potentials respectful of future market conditions and regional traditional wood industry demands so as to minimize the risk of problem shifting.

In Chapters 2, 3, and 5, it was found that current demands for road transportation fuel in Nordic regions are not compatible with "sustainable" regional and national resource potentials, reinforcing the notion that biofuels can only be a single part of a broader portfolio consisting of other technological and policy solutions – including demand side management and efficiency measures – that will ultimately be required in efforts to realize more sustainable transport. Results of scenario analyses of Chapter 5 imply that gains in fuel efficiency due to improvements in vehicle technologies, reductions in fuel intensity due to mode switching and changes in consumption structure, and reductions in the overall demand for transport services – can be more effective climate change mitigation strategies than fossil fuel substitution at the macro level.

Exemplified using the case of Fischer-Tropsch diesel in Norway in Chapter 4, production costs inclusive of inherent technical risks likely to be associated with the novel technology were quantified as were the public costs of various subsidized deployment scenarios. It was found that for a public cost of $100/tonne-GHG avoided and an oil price of $97/bbl, financial risks to pioneering technology investors can be greatly minimized through a policy package that combines a low interest government loan with a price floor.

It cannot be robustly concluded in black and white terms from the analysis performed in Chapter 6 that boreal forest biofuels in Norway will offer sustained climate benefits over the next century. Findings suggest that, when biofuels are produced from forest biomass felled via clear-cut methods, benefits do occur in the short term due to rapid albedo changes, but over the longer term cumulative biogenic carbon emissions from production and use of biofuels could lead to additional climate impact. Realizing a long-term climate benefit will have to stem from more efficient fossil fuel substitution and/or faster recycling of carbon, the latter of which calls for a greater role by forest management. Over the short term it is critical that forest management strategies are developed in light of trade-offs between the carbon cycle and albedo. Short term albedo benefits will have to be weighed against those which serve to enhance the overall productivity of the forest carbon sink over the longer term.

Research Outlook

The environmental systems analytic tools applied in this thesis were perceived most relevant at the time of their application for answering the main research questions. Attributional LCA is a structured method for comparing various forest-biofuel products and for learning about areas of environmental improvement in the product system and, whether hybrid or process-based, formed the backbone of most of the analyses. However, the inclusion of albedo and the explicit representation of time in climate impact assessment in Chaper 6 showed that the climate benefits/impacts of the alternative transport system based on forest biofuels occured at different points in time, calling into question the effectiveness of unit-based frameworks like LCA that rely soley on time-integrated emission metrics.

Initial research efforts by Cherubini et al. (2011) indicate that it may be possible to overcome the time issue in unit-based impact assessments of biofuels by developing and standardizing region- and species-specific characterization factors for biomass-based GHG emission. However, this imposes a requirement on the analyzer to be familiar with the embedded growth rates used in the development of these species-specific characterization factors, particularly if one has the goal of employing LCA to compare biofuel product alternatives produced in different regions. Further, strong assumptions are needed in cases where the origin or composition of the biomass feedstock inputs into a biofuel production process is not entirely known.

Efforts by Muñoz et al. (Muñoz, Campra, & Fernández-Alba, 2010) exemplify how albedo changes may be integrated in LCA-type frameworks. This is rather straightforward when land use imposes a permanent albedo change; however, for forest biofuels the albedo impacts cannot be normalized because the albedo change in not permanent. Following harvest, whether at a single tree or at stand level, the albedo change will be temporary, and its "decay" profile will depend on the time evolution of a variety of physical factors that are affected by local climate and human disturbance variables.

The difficulties of including these land use related modeling aspects call into question the utility of "fixed coefficient" analytic frameworks like LCA in stand alone applications for answering attributional-type questions about the climate impacts of forest-based biofuels. Approaches that integrate detailed land surface modeling with the life cycle inventory analysis procedure of LCA, radiative forcing analysis, and the analysis of shadow (reference) scenarios can be a more informative approach when the question or problem involves a fundamental shift in the way forests are utilized in product systems like "next generation" biofuels. The urgency of climate change mitigation and the need to deploy "beneficial" biofuels call for the immediate implementation of specific land use management strategies whose identification will likely require broader assessment frameworks such as the type employed in Chapter 6.

Identifying these specific management strategies and development opportunities – in addition to the screening of other near-term forest biofuel production technologies not covered in this thesis – ought to be the subject of future research efforts. Given the onset of peak oil and the growing surplus of boreal forest biomass in Nordic regions, it makes good sense to maintain multidisciplinary research efforts throughout the region. If unit-based tools like LCA are to be used for more informed decision making about forest biofuels, they ought to be balanced and used in conjunction with relevant frameworks capable of answering dynamic land use questions both at the single project and landscape levels.

References

Allenby, B. (1999). Industrial Ecology. New Jersey: Prentice-Hall, Inc.

Ardent, F., Beccali, M., & Cellura, M. (2009). Chapter 22: Application of the IO Methodology to the Energy and Environmental Analysis of a Regional Context. In S. Suh (Ed.), Handbook of Input-Output Economics in Industrial Ecology (pp. 435-457). Dordrecht/Heidelberg/London/New York: Springer.

Bala, G., Caldeira, K., Wickett, M., Phillips, T. J., Lobell, D. B., Delire, C., & Mirin, A. (2007). Combined climate and carbon-cycle effects of large-scale deforestation. PNAS, 104(16), 6550-6555.

Bare, J. C., Norris, G. A., Pennington, D. W., & McKone, T. E. (2003). TRACI, the tool for the reduction and assessment of chemical and other environmentral impacts. Journal of Industrial Ecology, 6, 49-78.

Bartis, J. T., Camm, F., & Ortiz, D. S. (2008). Producing liquid fuels from coal: Prospects and policy issues. Santa Monica, CA: RAND Corporation.

Baumann, Henrikke. (1998). Life cycle assessment and decision making: theories and practices. PhD, Chalmers University of Technology, Göteborg.

Baumann, Henrikke, & Tillman, Anne-Marie. (2004). The Hitch Hiker's Guide to LCA: An orientation in life cycle assessment methodology and application. Lund: Studentlitteratur.

Becker, E. (2002). Transformation of Social and Ecological issues into Transdisciplinary Research Knowledge for Sustainable Development: An Insight into the Encyclopaedia of Life Support Systems (pp. 949-963). Paris: UNESCO/EOLSS Publishers.

Berk, J., & DeMarzo, P. (2007). Corporate Finance. Boston: Pearson Education, Inc.

Berntsen, T., & Fuglestvedt, J. S. (2008). Global temperature responses to current emissions from the transport sectors. PNAS, 105(49), 19154-19159.

Betts, R. (2007). Implications of land ecosystem-atmosphere interactions for strategies for climate change adaptation and mitigation. Tellus B, 59(3), 602-615.

Betts, R. A. (2000). Offset of the potential carbon sink from boreal forestation by decreases in surface albedo. Nature, 408, 187-190.

Bonan, G. B. (2008). Forests and Climate Change: Forcings, Feedbacks, and the Climate Benefits of Forests. Science, 320, 1444-1449.

Bonan, G. B., & Pollard, D. (1992). Effects of boreal forest vegetation on global climate. Nature, 359(6397), 716-718.

Börjeson, L., Höjer, M., Dreborg, K.-H., Ekvall, T., & Finnveden, G. (2006). Scenario types and techniques: Towards a user's guide. Futures, 38(7), 723-739.

Borken-Kleefeld, J., Berntsen, T., & Fuglestvedt, J. S. (2010). Specific Climate Impact of Passenger and Freight Transport. Environmental Science & Technology, 44, 5700-5706.

Bringezu, S., Schütz, H., O'Brien, M., Kauppi, L., Howarth, R.W., & NcNeely, J. (2009). Towards Sustainable Production and Use of Resources: Assessing Biofuels. Paris: United Nations Environment Programme (UNEP), International Panel for Sustainable Resource Management (IPSRM).

Bruce, A., Lyall, C., Tait, J., & Williams, R. (2004). Interdisciplinary integration in Europe: the case of the Fifth Framework programme. Futures, 36(4), 457-470.

Camm, F., Bartis, J. T., & Bushman, C. J. (2008). Federal financial incentives to induce early experience producing unconventional liquid fuels. Santa Monica, CA: RAND Corporation.

Chalmers University of Technology. (2011). Accessed online at: http://www.chalmers.se/ee/EN/research/research-divisions/environmental-systems.

Cherubini, F., Peters, G. P., Berntsen, T., Strømman, A. H., & Hertwich, E. G. (2011). CO2 emissions from biomass combustion for bioenergy: atmospheric decay and contribution to global warming. Global Change Biology Bioenergy, doi: 10.1111/j.1757-1707.2011.0102.x.

Cherubini, F., & Strømman, A. H. (2010). Life cycle assessment of bioenergy systems: State of the art and future challenges. Bioresource Technology, 102(2), 437-451.

Cruetzig, F. S., & Kammen, Daniel M. (2010). Chapter 26: Getting the carbon out of transportation fuels. In H.-J. Schellenhuber, M. Molina, N. Stern, V. Huber & S. Kadner (Eds.), Global Sustainability: A Nobel Cause (pp. 307-318). Cambridge: Cambridge University Press.

Crutzen, P. J., Mosier, A. R., Smith, K. A., & Winiwarter, W. (2008). N2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. Atmospheric Chemistry and Physics, 8(2), 389-395.

Curran, M. A., Mann, M., & Norris, G. (2005). The international workshop on electricity data for life cycle inventories. Journal of Cleaner Production, 13(8), 853-862.

Denver Business Journal (Producer). (2011). Range Fuels closes biorefinery, seeks more funding. Accessed April 4, 2011 at: http://www.bizjournals.com/denver/print-edition/2011/01/28/range-fuels-closes-biorefinery-seeks.html.

Deutmeyer, M.W. (2010). Securing reliable long term solid biomass supply for large scale users Lyon, France: Presentation at 18th European Biomass Conference and Exhibition, May 6.

DG TREN. (2009). EU energy and transport in figures: Statistical pocketbook (pp. 232). Luxembourg: Directorate General for Energy and Transport (DG TREN) of the European Commission.

Ecoinvent. (2009). Ecoinvent v. 2.1 Database. from Swiss Centre for Life Cycle Inventories

Ehrlich, P. R., & Holdren, J. P. (1969). Population and Panaceas -- A Technological Perspective. Bioscience, 19, 1065-1071.

Ekvall, T. (2002). Limitations of Consequential LCA. California: International Conference on Life Cycle Assessment and Life Cycle Management, 20-25th May. Availbable at: http://www.lcacenter.org/lca-lcm/session-methods.html.

Ekvall, T., Tillman, Anne-Marie, & Molander, S. (2005). Normative ethics and methodology for life cycle assessment. Journal of Cleaner Production, 13, 1225-1234.

Ekvall, T., & Weidema, B. P. (2004). System Boundaries and Input Data in Consequential Life Cycle Inventory Analysis. International Journal of Life Cycle Assessment, 9(3), 161-171.

European Commission. (2009). A sustainable future for transport: Towards an integrated, technology-led and user friendly system. Brussels: Commission of the European Communities.

Faber, A., Idenburg, A., & Wilting, H. (2007). Exploring techno-economic scenarios in an input-output model. Futures, 39, 16-37.

FAO. (2007). Forests and energy in OECD countries. In W. E. Mabee & J. N. Saddler (Eds.): Forests and Energy Working Paper, United Nations Food and Agricultural Organization (FAO).

Farrell, A. E., Plevin, R. J., Turner, B. T., Jones, A. D., O'Hare, M., & Kammen, Daniel M. (2006). Ethanol Can Contribute to Energy and Environmental Goals. Science, 311(5760), 506-508.

Finnish Parliament. (2001). National Climate Strategy Finland -- Government Report to Parliament. Accessed at: http://julkaisurekisteri.ktm.fi/ktm_jur/ktmjur.nsf/All/CCDA20C55C9000A7C2256A6400218C F2/$file/National%20Climate%20Strategy%20Finland_2001.pdf. Helsinki.

Finnveden, G., Hauschild, M. Z., Ekvall, T., Guinée, J. B., Heijungs, R., Hellweg, S., . . . Suh, S. (2009). Recent developments in Life Cycle Assessment. Journal of Environmental Management, 91, 1-21.

Finnveden, G., & Moberg, Å. (2005). Environmental systems analysis tools -- an overview. Journal of Cleaner Production, 13, 1165-1173.

Frischknecht, R. (1998). Life Cycle Inventory Analysis for Decision-Making: Scope-dependent Inventory System Models and Context-specific Joint Production Allocation. PhD, ETH, Zürich.

Fuglestvedt, J. S., Shine, K. P., Berntsen, T., Cook, J., Lee, D.S., Stenke, A., . . . Waitz, I.A. (2009). Transport impacts on atmosphere and climate: Metrics. Atmospheric Environment, 44(37), 4648-4677.

Grübler, A., Nakicenovic, N., & Victor, D. G. (1999a). Dynamics of energy technologies and global change. Energy Policy, 27, 247-280.

Grübler, A., Nakicenovic, N., & Victor, D. G. (1999b). Modeling Technological Change: Implications for the Global Environment. Annual Review of Energy Environment, 24, 545-569.

Guinée, J. B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., Koning, A. de, . . . Huijbregts, M. A. J. (2002). Handbook on life cycle assessment. Operation guide to the ISO Standards. I: LCA in perspective. IIa: Guide. IIb: Operational annex. III: Scientific background. Dordrecht: Kluwer Academic Publishers.

Heinrich, A. B. (2010). International reference life cycle data system handbook. International Journal of Life Cycle Assessment, 15, 524-525.

Hettelingh, J.-P., Posch, M., De Smet, P. A. M., & Downing, R. J. (1995). The use of critical loads in emission reduction agreements in Europe. Water, Air, and Soil Pollution, 85, 2381-2388.

Hill, J., Nelson, E., Tilman, D., Polasky, S., & Tiffany, D. (2006). Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. PNAS, 103(30), 11206-11210.

Hill, J., Polasky, J., Nelson, E., Tilman, D., Huo, H., Ludwig, L., . . . Bonta, D. (2009). Climate Change and Health Costs of Air Emissions from Biofuels and Gasoline. Proceedings of the National Academy of Sciences, 106(6), 2077-2082.

Hofstetter, P. (1998). Perspectives in life cycle impact assessment: A structured approach to combine models of the technosphere, ecosphere, and valuesphere. Boston: Kluwer Academic Publishers.

Hofstetter, P., Bare, J. C., Hammitt, J. K., Murphy, P. A., & Rice, G. E. (2002). Tools for Compartive Analysis of Alternatives: Competing or Complementary Perspectives? Risk Analysis, 22(5), 833-851.

Höjer, M., Ahlroth, S., Dreborg, K.-H., Ekvall, T., Finnveden, G., Hjelm, O., . . . Palm, V. (2008). Scenarios in selected toos for environmental systems analysis. Journal of Cleaner Production, 16, 1958-1970.

Hojer, M., Ahlroth, S., Dreborg, K. H., Ekvall, T., Finnveden, G., Hjelm, O., . . . Palm, V. (2008). Scenarios in selected tools for environmental systems analysis. [Article]. Journal of Cleaner Production, 16(18), 1958-1970. doi: 10.1016/j.jclepro.2008.01.008

Hourcade, J. C., Jaccard, M., Bataille, C., & Ghersi, F. (2006). Introduction to the Special Issue of the energy journal. The energy journal special issue: Hybrid modelling of energy environment policies - reconciling bottom-up and top-down.

Huijbregts, M. A. J., Gilijamse, W., Ragas, A. M. J., & Reijnders, L. (2003). Evaluating uncertainty in environmental life-cycle assessment. A case study comparing two insulation options for a dutch one-family dwelling. Environmental Science & Technology, 37(11), 2600-2608.

Huijbregts, M. A. J., Schöpp, W., Verkuijlen, E., Heijungs, R., & Reijnders, L. (2000). Spatially explicit characterization of acidifying and eutrophying air pollution in life-cycle assessment. Journal of Industrial Ecology, 4, 75-92.

IEA. (2010). Sustainable Production of Second-Generation Biofuels. In A. Eisentraut (Ed.), Information Paper. Paris: OECD/IEA.

ISO 14040:2006. Environmental management -- Life cycle assessment -- Principles and framework. International Organisation for Standardisation (ISO), Geneve.

ISO 14044:2006. Environmental management -- Life cycle assessment -- Requirements and guidelines. International Organisation for Standardisation (ISO), Geneve.

Jabbar, M. A., Saleem, M., & Li-Pun, H. (2001). Evolution Toward Transdisciplinarity in Technology and Resource Management Research: The Case of a Project in Ethiopia. In J. T. Klein, W. Grossenbacher-Mansuy, R. Häberli, A. Bill, R. W. Scholz & M. Welti (Eds.), Transdisciplinarity: Joint Problem Solving among Science, Technology, and Society: An Effective Way for Managing Complexity. Basel, Boston, Berlin: Birkhäuser.

Jackson, R. B., Randerson, J. T., Canadell, J. G., Anderson, R. G., Avissar, R., Baldocchi, D. D., . . . Pataki, D., E. (2008). Protecting climate with forests. Environmental Research Letters, 3, 044006 (044005pp).

Jacobson, M. Z. (2007). Effects of Ethanol (E85) versus Gasoline Vehicles on Cancer and Mortality in the United States. Environmental Science & Technology, 41, 4150-4157.

Jacoby, H. D., Reilly, J. M., McFarland, J. R., & Paltsev, S. (2006). Technology and technical change in the MIT EPPA model. Energy Economics, 28, 610-631.

Joshi, S. (2000). Product environmental life-cycle assessment using input-output technique. Journal of Industrial Ecology, 3(2-3), 95-120.

Kammen, Daniel M., Farrell, A. E., Plevin, R. J., & Jones, A. D. (2008). Energy and greenhouse impacts of biofuels: A framework for analysis Biofuels: Linking Support to Performance (pp. 41-74). Paris: OECD/ITF.

Kauppi, P., & Saikku, L. (2009). Biofuels: Forests and Carbon. Science, 326(5958), 1345.

Kim, H., Kim, S., & Dale, B. E. (2009). Biofuels, Land Use Change, and Greenhouse Gas Emissions: Some Unexplored Variables. Environmental Science & Technology, 43, 961-967.

Klein, J. T. (2004). Interdisciplinarity and complexity: an evolving relationship. E:CO, 6(1-2), 2-10.

Koellner, T., & Scholz, R. (2007). Assessment of the land use impacts on the natural environment. Part 1: An analytical framework for pure land occupation and land use change. International Journal of Life Cycle Assessment, 12, 16-23.

Koellner, T., & Scholz, R. (2008). Assessment of the land use impacts on the natural environment. Part 2: Generic characterization factors for local species diversity in central Europe. International Journal of Life Cycle Assessment, 13, 32-48.

Leiden University. (2001). Life Cycle Assessment -- An Operational Guide to ISO Standards, Version 2. Leiden, The Netherlands: Centre of Environmental Science (CML), Leiden University.

Lenzen, M., Pade, L.-L., & Munksgaard, J. (2004). CO2 Multiplier in Multi-Region Input-Output Models. Economic Systems Research, 16(4), 391-412.

Lindeijer, E., Müller-Wenk, R., & Steen, B. (2002). Impact assessment of resources and land use. In H. A. Udo de Haes, G. Finnveden, M. Goedkoop, M. Z. Hauschild, E. G. Hertwich, P. Hofstetter, O. Jolliet, W. Klöpffer, W. Krewitt & E. Lindeijer (Eds.), Life-Cycle Impact Assessment: Striving Towards Best Practice. Pensacola: SETAC Press.

Lloyd, S. M., & Ries, R. (2007). Characterizing, Propogating, and Analyzing Uncertainty in Life Cycle Assessment. Journal of Industrial Ecology, 11(1), 161-179.

Lowe, P., & Phillipson, J. (2009). Barriers to research collaboration across disciplines: scientific paradigms and institutional practices. Environment and Planning A, 41, 1171-1184.

Luo, L., van der Voet, E., Huppes, G., & Udo de Haes, H. A. (2009). Allocation issues in LCA methodology: A case study of corn stover-based fuel ethanol. International Journal of Life Cycle Assessment, 14, 529-539.

Lyons, E., Jin, Y., & Randerson, J. T. (2008). Changes in surface albedo after fire in boreal forest ecosystems of interior Alaska assessed using MODIS satellite observations. Journal of Geophysical Research, 13, G02012.

Marland, G., & Marland, S. (1992). Should we store carbon in trees? Water, air, and soil pollution, 64(1-2), 181-195.

Marland, G., Pielke, R. A., Apps, M., Avissar, R., Betts, R. A., Davis, K. J., . . . Xue, Y. (2003). The climatic impacts of land surface change and carbon management, and the implications for climate-change mitigation policy. Climate Policy, 3, 149-157.

McFarland, J. R., Reilly, J. M., & Herzog, H. J. (2004). Representing energy technologies in top-down economic models using bottom-up information. Energy Economics, 26, 685-707.

McGuire, A. David, Chapin, F.S., Walsh, John E., & Wirth, Christian. (2006). Integrated Regional Changes in Arctic Climate Feedbacks: Implications for the Global Climate System*. Annual Review of Environment and Resources, 31(1), 61-91. doi: doi:10.1146/annurev.energy.31.020105.100253

McKechnie, J., Colombo, S., Chen, J., Mabee, W. E., & MacLean, H. L. (2011). Forest Bioenergy or Forest Carbon? Assessing Trade-Offs in Greenhouse Gas Mitigation with Wood-Based Fuels. Environmental Science & Technology, 45, 789-795.

Meinshausen, M., Meinshausen, N., Hare, W., Raper, S. C. B., Frieler, K., Knutti, R., . . . Allen, M. R. (2009). Greenhouse-gas emissions targets for limiting global warming to 2 degrees C. Nature, 458, 1158-1164.

Merrow, W.E., Phillips, K.E., & Myers, C.W. (1981). Understanding cost growth and performance shortfalls in pioneer process plants. USA: The Rand Corporation.

Michelsen, O. (2008). Assessment of land use impact on biodiversity - Proposal of a new methodology exemplified with forestry operations in Norway. International Journal of Life Cycle Assessment, 13, 22-31.

Mila i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., . . . Rydgren, B. (2007). Key elements in a framework for land use impact assessment within LCA. International Journal of Life Cycle Assessment, 12, 5-15.

Miller, R. E. (1969). Interregional feedbacks in input-output models: some experimental resutls. Western Economic Journal, 7, 57-70.

Morse, W. C., Nielsen-Pincus, M., Force, J., & Wulfhorst, J. (2007). Bridges and barriers to developing and conducting interdisciplinary graduate-student team research. Ecology and Society, 12(2), 8.

Muñoz, I., Campra, P., & Fernández-Alba, A. R. (2010). Including CO2-emission equivalence of changes in land surface albedo in life cycle assessment. Methodology and case study on greenhouse agriculture. International Journal of Life Cycle Assessment, 15, 672-681.

Murray, J., Wood, R., & Lenzen, M. (2010). Chapter 2: Input-Output Analysis -- Strenghts and Limitations. In J. Murray & R. Wood (Eds.), The Sustainability Practioner's Guide to Input-Output Analysis (pp. 23-38). Champlaign, Illinois: Common Ground Publishing, LLC.

Mutel, C. L., & Hellweg, S. (2009). Regionalized Life Cycle Assessment: Computational Methodology and Application to Inventory Databases. Environmental Science & Technology, 43, 5797-5803.

National Academy of Sciences, National Academy of Engineering, & Institute of Medicine of the National Academies. (2005). Facilitating Interdisciplinary Research. Washington, D. C.: The National Academies Press.

Norway and Sweden agree on a common market for green certificates (2010).

Ohlrogge, John, Allen, Doug, Berguson, Bill, DellaPenna, Dean, Shachar-Hill, Yair, & Stymne, Sten. (2009). Driving on Biomass. Science, 324(5930), 1019-1020. doi: 10.1126/science.1171740

Olsen, S. I., Christensen, F. M., Hauschild, M. Z., Pedersen, F., Larsen, H. F., & Tørsløv, J. (2001). Life cycle impact assessment and risk assessment of chemicals -- a methodological comparison. Environmental Impact Assessment Review, 21, 385-404.

Owens, J. W. (1997a). Life-Cycle Assessment in Relation to Risk Assessment: An Evolving Perspective. Risk Analysis, 17(3), 359-365.

Owens, J. W. (1997b). Life-Cycle Assessments: Constraints on Moving from Inventory to Impact Assessment. Journal of Industrial Ecology, 1(1), 37-50.

Pearce, D., Atkinson, G., & Mourato, S. (2006). Cost-Benefit Analysis and the Environment. Paris: OECD Publishing.

Peters, G. P., & Hertwich, E. G. (2009). Chapter 38: The Application of Multi-regional Input-Output Analysis to Industrial Ecology. In S. Suh (Ed.), Handbook of Input-Output Economics in Industrial Ecology (pp. 847-863). Dordrecht/Heidelberg/London/New York: Springer.

Peters, M. S., Timmerhaus, K. D., & West, R. E. (2003). Plant Design and Economics for Chemical Engineers, 5th Ed. (Vol. 5th edition): McGraw - Hill.

Pielke Sr., R. A., Marland, G., Betts, R. A., Chase, T. N., Eastman, J. L., Niles, J. O., . . . Running, S. W. (2002). The influence of land-use change and landscape dynamics on the climate system: relevance to climate-change policy beyond the radiative effect of greenhouse gases. Phil. Trans. R. Soc. Lond. A, 360, 1705-1719.

Pimentel, D., Marklein, A., Toth, M. A., Karpoff, M., Paul, G. S., McCormack, R., . . . Krueger, T. (2008). Biofuel Impacts on World Food Supply: Use of Fossil Fuel, Land, and Water Resources. Energies, 1(2), 41-78.

Pimentel, D., & Pimentel, M. (2008). Food, Energy, and Society (3rd ed.). Boca Raton, FL: CRC.

Posch, M., Hettelingh, J.-P., & De Smet, P. A. M. (2001). Characterization of critical load exceedances in Europe. Water, Air, and Soil Pollution, 130, 1139-1144.

Potting, J., & Hauschild, M. Z. (1997). Predicted environmental impact and expected occurrence of actual environmental impact. Part 2: Spatial differentiation in life-cycle assessment via the site-dependent characterization of environmental impact from emissions. International Journal of Life Cycle Assessment, 2, 209-216.

Rabl, A., Benoist, A., Dron, D., Peuportier, B., Spadaro, J. V., & Zoughaib, A. (2007). How to Account CO_2 Emissions from Biomass in an LCA. International Journal of Life Cycle Assessment, 12(5), 281.

Randerson, J. T., Liu, H., Flanner, M. G., Chambers, S. D., Jin, Y., Hess, P. G., . . . Zender, C. S. (2006). The Impact of Boreal Forest Fire on Climate Warming. Science, 314, 1130-1132.

Ribeiro, S. K., Kobayashi, S., Beuthe, M., Gasca, J., Greene, D., Lee, D.S., . . . Zhou, P.J. (2007). Transport and Its Infrastructure. In O.R. Davidson [B. Metz, P.R. Bosch, R. Dave, L.A. Meyer] (Ed.), 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, United Kingdom and New York, NY, USA.

Robinson, J. (2004). Squaring the circle: some thoughts on the idea of sustainable development. Ecological Economics, 48(4), 369-384.

Robinson, J. (2008). Being undisciplined: Transgressions and intersections in academia and beyond. Futures, 40, 70-86.

Roeckner, E., Giorgetta, M. A., Crueger, T., Esch, M., & Pongratz, J. (2011). Historical and future anthropogenic emission pathways derived from coupled climate-carbon cycle simulations. Climatic Change, 105(18), 91-108.

Rosenfield, P. L. (1992). The potential of transdisciplinary research for sustaining and extending linkages between the health and social sciences. Social Science & Medicine, 35(11), 1343-1357.

Sandén, Björn A., & Karlström, M. (2007). Positive and negative feedback in consequential life-cycle assessment. Journal of Cleaner Production, 15(1469-1481).

Savage, N. (2011). Fuel Options: The ideal biofuel. Nature, 474(S9-S11), doi: 10.1038/1474S1009a.

Schafer, A., & Jacoby, H. D. (2003). Technology Detail in a Multi-Sector CGE Model: Transport Under Climate Policy, Report No. 101 MIT Joint Program on the Science and Policy of Global Change. Cambridge, MA: Center for Energy & Environmental Studies and Department of Geography & Environment, Boston University, and Joint Program on the Scince & Policy of Global Change, Massachusetts Institute of Technology (MIT).

Schlamadinger, B., Apps, M., Bohlin, F., Gustavsson, L., Jungmeier, G., Marland, G., . . . Savolainen, I. (1997). Towards a standard methodology for greenhouse gas balances of bioenergy systems in comparison with fossil energy systems. Biomass and Bioenergy, 13(6), 359-375.

Schlamadinger, B., & Marland, G. (1996a). Full fuel cycle carbon blances of bioenergy and forestry options. Energy Conservation and Management, 37(6-8), 813-818.

Schlamadinger, B., & Marland, G. (1996b). The Role of Forest and Bioenergy Strategies in the Global Carbon Cycle. Biomass and Bioenergy, 10(5/6), 275-300.

Searchinger, T., Heimlich, R., Houghton, R., Dong, F., Elobeid, A., Fabiosa, J., . . . Yu, T.-H. (2008). Use of U.S. Croplands for Biofuels Increases Greenhouse Gases through Emissions from Land Use Change. Science, 319(5867), 1238-1240.

Seppälä, J., Posch, M., Johansson, M., & Hettelingh, J.-P. (2006). Country-Dependent Characterization Factors for Acidification and Terrestrial Eutrophication Based on Accumulated Exceedance as an Impact Category Indicator. International Journal of Life Cycle Assessment, 11(6), 403-419.

Smeets, E.M.W., & Faaij, A.P.C. (2007). Bioenergy Potentials from Forestry in 2050. Climatic Change, 81, 353-390.

Stoeglehner, G., & Narodoslawsky, M. (2009). How sustainable are biofuels? Answers and further questions arising from an ecological footprint perspective. Bioresource Technology, 100, 3825-3830.

Strømman, Anders Hammer, Hertwich, E. G., & Duchin, Faye. (2009). Shifting Trade Patterns as a Means of Reducing Global Carbon Dioxide Emissions: A Multiobjective Analysis. Journal of Industrial Ecology, 13(1), 38-57.

Suh, S., & Huppes, G. (2002). Missing inventory compilation tool using extended input-output analysis. International Journal of Life Cycle Assessment, 7(3), 134-140.

Suh, S., & Huppes, G. (2005). Methods for Life Cycle Inventory of a product. Journal of Cleaner Production, 13, 687-697.

Suh, S., & Huppes, G. (2009). Methods in the Life Cycle Inventory of a Product. In S. Suh (Ed.), Handbook of Input-Output Economics in Industrial Ecology. Dordrecht/Heidelberg/London/New York: Springer.

Suh, S., & Kagawa, S. (2009). Chapter 3: Industrial Ecology and Input-Output Economics -- A Brief History. In S. Suh (Ed.), Handbook of Input-Output Economics in Industrial Ecology (pp. 43-58). Dordrecht/Heidelberg/London/New York: Springer.

Suh, S., Lenzen, M., Treloar, G.J., Hondo, H., Horvath, A., Huppes, G., . . . Norris, G. (2004). System Boundary Selection in Life Cycle Inventories Using Hybrid Approaches. Environmental Science and Technology, 38(657-664).

Suh, S., & Nakamura, S. (2007). Five Years in the Area of Input-Output and Hybrid LCA. International Journal of Life Cycle Assessment, 12(6), 351-352.

Swann, A. L., Fung, I. Y., Levis, S., Bonan, G. B., & Doney, S. C. (2010). Changes in Arctic vegetation amplify high-latitude warming through the greenhouse effect. PNAS, 107(4), 1295-1300.

Tainter, J. A. (1988). The Collapse of Complex Societies - New studies in archaeology. Cambridge: Cambridge University Press.

Tainter, J. A. (2000). Problem Solving: Complexity, History, Sustainability. Population and Environment, 22(1), 3-41.

Tapio, Petri. (2005). Towards a theory of decoupling: degrees of decoupling in the EU and the case of road traffic in Finland between 1970 and 2001. Transport Policy, 12(2), 137-151.

Thompson, M., Adams, D., & Johnson, K.N. (2009). The Albedo Effect and Forest Carbon Offset Design. Journal of Forestry, 107(8), 425-431.

Tilman, D., Socolow, R., Foley, J. A., Hill, J., Larson, E., Lynd, L., . . . Williams, R. (2009). Beneficial Biofuels -- The Food, Energy, and Environment Trilemma. Science, 325(270-272).

Tilman, David, Hill, J., & Lehman, C. (2006). Carbon-Negative Biofuels from Low-Input High-Diversity Grassland Biomass. Science, 314, 1598-1600.

TØI. (2008). Transportytelser i Norge 1946-2007 [Transport volumes in Norway 1946-2007]. In Arne Rideng & Liva Vågane (Eds.). Oslo: Institute of Transport Economics (TØI).

Treloar, G. (1997). Extracting embodied energy paths from input-output tables: Towards and input-output-based hybrid energy analysis method. Economic Systems Research, 9(4), 375-391.

Tukker, A., Eder, P., & Suh, S. (2006). Environmental impacts of products - Policy relevant information and data challenges. [Article]. Journal of Industrial Ecology, 10(3), 183-198.

Udo de Haes, H. A. (2006). How to approach land use in LCIA, or how to avoid the Cindarella effect? Comments on 'Key elements in a framework for land use impact assessment within LCA'. International Journal of Life Cycle Assessment, 11, 219-221.

Udo de Haes, H. A., Heijungs, R., Huppes, G., van der Voet, E., & Hettelingh, J.-P. (2000). Full Mode and Attribution Mode in Environmental Analysis. Journal of Industrial Ecology, 4(1), 45-57.

Udo de Haes, H. A., Heijungs, R., Suh, S., & Huppes, G. (2004). Three Strategies to Overcome the Limitations of Life-Cycle Assessment. Journal of Industrial Ecology, 8(3), 19-32.

Udo de Haes, H. A., Jolliet, O., Finnveden, G., Hauschild, M. Z., Krewitt, W., & Müller-Wenk, R. (1999). Best available practice regarding impact categories and category indicators in life cycle impact assessment: Background document for the second working group on life cycle impact assessment of SETAC-Europe (WAI-2). International Journal of Life Cycle Assessment, 4(2), 66-74.

UN FAO. (2010). Global Forest Resources Assessment 2010 FAO Forestry Paper 163. Rome: Food and Agriculture Organization of the United Nations (UN FAO).

van der Voet, E., Lifset, R. J., & Luo, L. (2010). Life-cycle assessment of biofuels, convergence and divergence. Biofuels 1(3), 435-449.

Vögele, S., Kuckshinrichs, W., & Markewitz, P. (2009). Chapter 17: A Hybrid IO Energy Model to Analyze CO2 Reduction Policies - A Case of Germany. In S. Suh (Ed.), Handbook of Input-Output Economics in Industrial Ecology (pp. 337-356). Dordrecht/Heidelberg/London/New York: Springer.

Wageningen University. (2011). Accessed at: http://www.esa.wur.nl/UK/

Weber, C. L., Hendrickson, C. T., & Matthews, H. S. (2010). Chapter 10: The Economic Input-Output Life Cycle Assessment (EIO-LCA) Model. In J. Murray & R. Wood (Eds.), The Sustainability Practitioner's Guide to Input-Output Analysis (pp. 125-143). Champaign, Illinois: Common Ground Publishing, LLC.

Wiedmann, T. (2009). A review of recent multi-region input-output models used for consumption-based emission and resource accounting. Ecological Economics, 69(2), 211-222.

Wilting, H. C. (1996). An energy perspective on economic activities. PhD, University of Gröningen, The Netherlands.

Wilting, Harry C., Faber, A., & Idenburg, A. M. (2008). Investigating new technologies in a scenario context: description and application of an input-output method. Journal of Cleaner Production, 16(S1), S102-S112.

Wrisberg, N., Udo de Haes, H. A., Bilitewski, B., Bringezu, S., Bro-Rasmussen, F., Clift, R., . . . Triebswetter, U. (2002). Part I: Demand and supply of environmental information. In N. Wrisberg, H. A. Udo de Haes, U. Triebswetter, P. Eder & R. Clift (Eds.), Analytical Tools for Environmental Design and Management in a Systems Perspective. Dordrecht/Boston/London: Kluwer Academic Publishers.

Appendix A: Supporting Information for Paper V

SUPPORTING INFORMATION for

Radiative Forcing Impacts of Boreal Forest Biofuels: A Scenario Study for Norway in Light of Albedo

*Ryan M. Bright[*1], Anders Hammer Strømman[1], Glen P. Peters[2]*

This file includes:

General Parameter Values

Materials and Methods

Additional Results Figures and Tables

References and Notes

General Parameters

The following parameter values are used in climate impact assessment:

Mass of atmosphere (M_a) = 5.1441 e+18 kg

Area of Earth's surface (A_E) = 5.10072 e+14 m^2

Molecular weight of air (M_{air}) = 28.97 kg/kmol

Molecular weight of CO_2 (M_{CO2}) = 44.009 kg/kmol [1]

Background CO_2 concentration (CO_2') = 378 ppmv [1]

Radiative efficiency of CO_2 (a_{CO2}) = 5.35*ln$((CO_2'+1)/(CO_2'))$ W/m^2/ppmv [1, 2]

Radiative efficiency of CO_2, mass (k_{CO2}) = a_{CO2}/(1e-6*M_{CO2}/M_{air}*M_a) W/m^2/kg [1]

Radiative efficiency of CH_4 mass (k_{CH4}) = 1.82 e-13 W/m^2/kg [1]

Lifetime CH_4 (τ_{CH4}) = 12 years [1]

Radiative efficiency of N_2O (k_{N2O}) = 3.88 e-13 (W/m^2/kg) [1]

Lifetime N_2O (τ_{N2O}) = 14 years [1]

Table S1. Parameters a_i and b_i of the impulse response function (IRF) for the decay of CO_2 adopted from [1].

	$i = 0$	$i = 1$	$i = 2$	$i = 3$
a_i (unitless)	0.217	0.259	0.338	0.186
b_i (years)		172.9	18.51	1.186

Emissions and Radiative Forcing CO_2

A primary supposition adopted in this paper is that all CO_2 is treated as being equal in the atmosphere and that carbon neutrality does not necessarily equate to climate neutrality. All CO_2 emissions, whether of fossil- or bio- origin, alter the carbon cycle and Earth's radiative balance and contribute to climate change. This contribution is quantified by means of cumulative radiative forcing, which is based on the atmospheric decay over time of a pulse emission. This decay is modeled using an updated CO_2 impulse response function (IRF) adopted by the IPCC [1] which is based on the Bern 2.5 Carbon Cycle model [3]. The CO_2 IRF takes the following analytical form, with its parameter values shown in Table S1:

$$y_{CO_2}(t) = a_0 + \sum_{i=1}^{i=3} a_i e^{\left(-\frac{t}{b_i}\right)} \tag{S1}$$

The value of this function at any time represents the fraction of the initial emission which remains in the atmosphere. A detailed description of this model can be found elsewhere [3, 4].

The time evolution of an instantaneous forcing ΔRF_{CO2} due to 1 kg pulse emission at time zero can be expressed as:

$$\Delta RF_{CO_2}(t) = k_{CO_2}\left(a_0 + \sum_{i=1}^{3} a_i \exp\left(\frac{-t}{b_i}\right)\right) \tag{S2}$$

where k_{CO2} is the radiative efficiency of CO_2 expressed per kg given a background concentration of 378ppmv (in $W/m^2/kg$). Concentration increases over time due to scenario emissions are neglected as this increase is negligible, thus k_{CO2} remains constant. Integrating ΔRF_{CO2} over t gives us the absolute global warming potential **AGWP** at time t due to a single kg pulse emission at $t=0$:

$$AGWP_{CO_2}(t) = \int_0^t \Delta RF_{CO_2}\, dt = \int_0^t k_{CO_2}\left(a_0 + \sum_{i=1}^{3} a_i \exp\left(\frac{-t}{b_i}\right)\right) dt \tag{S3}$$

Time-integrated radiative forcing (iRF) due to a series of annual pulse emissions over time associated with an emissions scenario, iRF^S_{CO2}, can be expressed as a convolution of $AGWP_{CO2}$ and annual CO_2 emission e_{CO2}:

$$iRF^S_{CO_2}(t) = \int_{t'=0}^{t'=t} e_{CO_2}(t')AGWP_{CO_2}(t-t')\,dt' = \int_{t'=0}^{t'=t} k_{co_2} e_{co_2}(t') y_{co_2}(t-t')\,dt' \tag{S4}$$

In this study we only make use of the marginal emission scenario, $e_{CO2}{}^{BF} - e_{CO2}{}^{FR}$, or Δe_{CO2} for "e_{CO2}".

N_2O and CH_4

For a non-CO_2 substance x considered in this study, we make use of IPCC radiative efficiency k_x and lifetime values τ_x [1] listed above together with an exponential decay function to derive the instantaneous forcing profile due to a 1 kg pulse emission:

$$\Delta RF_x(t) = k_x\left(1 - \exp\left(-\frac{t}{\tau_x}\right)\right) \tag{S5}$$

The AGWP of non-CO_2 substance "x" at time "t" due to a 1 kg pulse emission at time $t=0$ for (in W/m^2), can be written:

$$AGWP_x(t) = \int_0^t \Delta RF_x(t)\,dt = \int_0^t k_x\left(\exp\frac{-t}{\tau_x}\right) dt \tag{S6}$$

The iRF^S_x at time t due to a series of pulse emissions e_x in an emission scenario for non-CO_2 substance x is represented by a convolution of pulse emissions and AGWP:

$$iRF^S_x(t) = \int_{t'=0}^{t'=t} e_x(t')AGWP_x(t-t')\mathrm{d}t' = \int_{t'=0}^{t'=t} k_x e_x(t')y_x(t-t')\mathrm{d}t' \tag{S7}$$

Again, we only make use of the marginal emission scenario for substance "x".

Albedo and Radiative Forcing

The albedo data are based on the Terra and Aqua (Combined) MODIS BRDF/Albedo Model Parameter Product (MCD43A1) which are calculated using a solar zenith angle equal to the local solar noon and an optical depth of 0.2 [5]. BRDF is the "Bidirectional Reflectance Distribution Function" giving the reflectance of a target as a function of illumination geometry and viewing property. The BRDF depends on wavelength, and is determined by the structural and optical properties of the surface such as shadow-casting, multiple scattering, mutual shadowing, transmission, reflection, absorption and emission by surface elements, facet orientation distribution, and facet density (for more information, see: http://www-modis.bu.edu/brdf/userguide/index.html).

We apply monthly average cloud-cleared total-sky surface albedo ("α") data spanning Jan. 2001 – Dec. 2009 together with Fu-Liou [6] and cloud/climate data [7] to derive single values of the 24-hr. average net shortwave radiation flux at the top of the atmosphere for each mid-month Julian day, ΔSW_{TOA}, for each IGBP land use type in each of the five primary logging regions throughout Norway. IGBP is the International Geosphere-Biosphere Programme providing a standard framework for the systematic classification of land surface typology according to shared physical, chemical, and biological features. The MODIS BRDF/Albedo product (MCD43A1) is integrated with MODIS Land Cover Product (MOD12, Collection 5) which is based on IGBP Classification Type 1.

The mid-monthly 24-hr. average instantaneous albedo-ΔSW_{TOA} values are then weighted according to the relative volume of annual logging activity that occurred in each logging region in the year 2009 in order to derive one annual *geographically-weighted* 24-hr. average local value for each land surface type, or $\Delta SW_{TOA}^{(Year,\ IGBP)}$:

$$\Delta SW_{TOA}^{Year,IGBP} = \sum_{n=1}^{5} R_n \left(\frac{\sum_{month=1}^{12} \Delta SW_{TOA}^{Month,IGBP}}{12} \right) \tag{S8}$$

where $\boldsymbol{R}$ represents the regional weighting factors shown in Figure S1 and Figure 1 and $\Delta SW_{TOA}^{(Month,\ IGBP)}$ is the monthly 24-hr. ΔSW_{TOA} flux over the corresponding IGBP surface type.

The steps are repeated using two times the standard deviation of the monthly albedo at each site to derive new, geographically-weighted $\Delta SW_{TOA}^{(Year,\ IGBP)}$ values. The higher bounds for *Needleleaf Evergreen* and *Mixed Forest* IGBP types are used together with the lower bound for *Open Shrub*, and vice versa, to create two geographically-weighted local $\Delta SW_{TOA}^{(Year,\ IGBP)}$ uncertainty scenarios. These values are presented in Table 1 of the main manuscript.
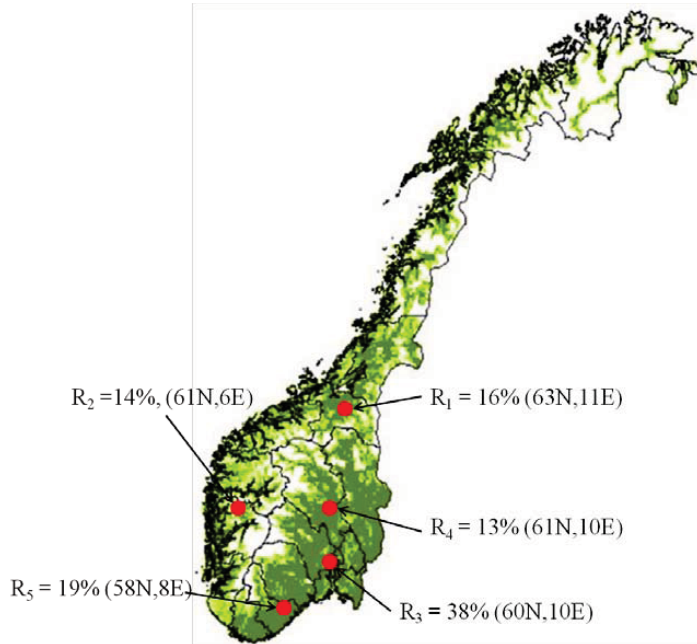
Figure S1. Latitude and longitude coordinates of sample regions and the corresponding share of logging activity, "*R*", used in geographic weighting. Dark green refers to areas in Norway classified as heavily managed productive forests. Figure is adapted from ref [8].

Radiative Forcing from Albedo Change

It is possible to estimate a local ΔSW_{TOA} flux time series in each scenario because our land use model accounts for clear-cut area plus area under growing stock for each species i and each age a within productive forests over time t for both of our scenarios. Each species i falls into one of the two IGBP classifications: "*(1) Needleleaf Evergreen Forests*" as a proxy for spruce/pine-dominant areas and "*(5) Mixed Forests*" for birch-dominant areas. For any given time step with increment of one year, $\Delta SW_{TOA}^{(Year,\ IGBP)}$ is assumed to represent the maximum annual local ΔSW_{TOA} over harvestable productive forest areas, or $\Delta SW^{Max}\ (i)$.

In order to estimate $\Delta SW_{TOA}^{(Year,\ IGBP)}$ as a function of both forest type i and age a, we need to introduce a variable that corresponds to the age of $\Delta SW^{Max}\ (i)$. Empirical albedo measurement data of boreal forests types as a function of age are not available for Norway,

thus key assumptions relating important physical properties of managed forests to local albedo-ΔSW_{TOA} have to be made. Previous studies [9, 10] indicate that the evolution of canopy closure and leaf area index (LAI) are important driving factors in the evolution of albedo in managed boreal forests of uniform species and age. Additionally, [9] show that the evolution of these parameters are both linear and will eventually saturate at a clear age, depending on species type and site productivity, shown in Table S2.

Table S2. Albedo-ΔSW^{Max} (i) ages (*a*) are based on the combined average saturation age of canopy closure and LAI, adapted from ref. [9]. *FB* = *"Fertile Birch"*; *IFB* = *"Infertile Birch"*; *FP* = *"Fertile Pine"*; *IFP* = *"Infertile Pine"*; *FS* = *"Fertile Spruce"*.

	FB	IFB	FP	IFP	FS
Canopy Closure Saturation Age	20	20	20	40	30
LAI Saturation Age	40	70	20	40	50
Mean Saturation Age , LAI + Canopy Closure	30	45	20	40	40
Mean Saturation Age, Species aggregate	38		30		40
	i, Mixed Forests		*i*, Evergreen Needleleaf Forests		
a, Mean, IGBP aggregate	38		35		
a, Higher bound, IGBP aggregate	45		40		
a, Lower bound, IGBP aggregate	30		30		

The combined LAI + canopy closure saturation ages shown in Table S2 are used as a proxy for local annual ΔSW^{Max} (i), henceforth referred to as $\boldsymbol{\tau}$. Albedo and ΔSW_{TOA} scale linearly with each other and with age up until $\boldsymbol{\tau}$, shown conceptually in Figure S2. Following ref. [9], LAI and canopy are shown to stabilize and flatten after the saturation ages in managed forests, thus we assume albedo does as well; thus, for all ages above $\boldsymbol{\tau}$, ΔSW^{Max} (i) is applied.
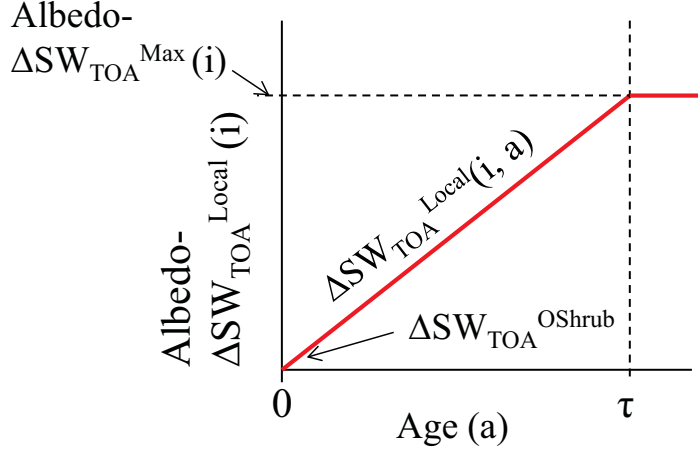
Figure S2. Linear relationship between age a and local albedo-$\Delta SW^{Max}(i)$ up until τ. Age values for τ are shown in Table S2.

For younger-aged species, the difference between the annual mean local net flux over clear-cut areas -- $\Delta SW_{TOA}^{(Year, OShrub)}$ – and $\Delta SW^{Max}(i)$ is adjusted linearly for ages less than τ:

$$\Delta SW_{TOA}^{Local}(i,a) = \begin{cases} \Delta SW_{TOA}^{OShrub} + \dfrac{a}{\tau}\left(\Delta SW_{TOA}^{Max}(i) - \Delta SW_{TOA}^{OShrub}\right) & \forall a < \tau \\ \Delta SW_{TOA}^{Max}(i) & \forall a \geq \tau \end{cases} \tag{S9}$$

By knowing the age distribution by species and IGBP type within productive forests in Norway $A(i,a)$ in any given time step t for any scenario S, it is possible to convert local net shortwave fluxes at the top of the atmosphere into global fluxes:

$$\Delta SW_{TOA}^{S,Global}(i,a,t) = A_E^{-1}\left(\Delta SW_{TOA}^{Local}(i,a)A(i,a,t)\right) \tag{S10}$$

where A_E represents the surface area of the earth and $\Delta SW^{Local}(i,a)$ is the mean annual local flux of species i of age a at time t when t and a are one year increments. Summing the annual mean global fluxes for all species and ages gives us the total flux in a single time step and for a given scenario. The difference in the global mean net annual flux at any time step t between

scenarios gives us the global radiative forcing change due to albedo (α) changes on productive forest areas:

$$\Delta RF_\alpha^{Global}(t) = \Delta SW_{TOA}^{BF,Global}(t) - \Delta SW_{TOA}^{FR,Global}(t) \tag{S11}$$

Expressing Cumulative Annual Albedo Forcing as Annual CO2-eq. Emissions

Because of its widespread implementation, we express the contribution of cumulative annual forcing changes due to albedo changes in forests over time in terms of annual pulse CO_2-equivalent emissions. For albedo, we have already determined a $\Delta RF_\alpha^{Global}(t)$ profile (Eq. (S11)) due to marginal albedo forcing changes in forests over time. Solving for a single pulse CO_2-equivalent emission at time $t=0$ that gives the same $\Delta RF_\alpha^{Global}(t)$ profile would be easy; however, we need to solve for a *time series* of pulse CO_2-equivalent emissions that yields the identical changing radiative forcing profile over time. This requires us to solve for *e(t)* in the following convolution of pulse CO_2 emissions and a CO_2 decay profile, where *y(t)* represents the CO_2 pulse-response function (Eq. (S1) above):

$$\Delta RF_\alpha^{Global}(t) = \int_{t'=0}^{t'=t} k_{CO_2} e_{CO_2}(t') y_{CO_2}(t-t') dt' \tag{S12}$$

We do not solve for *e(t)* analytically but instead we solve for it discretely on an annual emission basis by rewriting Eq. (S12) to:

$$\Delta RF_\alpha^{Global}(t) = \left(\sum_{t'=0}^{t'=t} k_{CO_2} e_{CO_2}(t') y_{CO_2}(t-t') \right) \tag{S13}$$

Solving for the discrete annual emission scenario is made easier through implementation of matrix algebra:

$$\begin{pmatrix} \Delta RF_\alpha^{Global}(t_0) \\ \vdots \\ \vdots \\ \vdots \\ \Delta RF_\alpha^{Global}(t_n) \end{pmatrix} = \begin{pmatrix} y(t_0) \\ y(t_1), y(t_0) \\ y(t_2), y(t_1), y(t_0) \\ \vdots \quad\quad\quad \ddots \\ y(t_n) \quad \cdots \quad\quad y(t_0) \end{pmatrix} \begin{pmatrix} e(t_0) \\ \vdots \\ \vdots \\ \vdots \\ e(t_n) \end{pmatrix} \odot \begin{pmatrix} k_{CO_2} \\ \vdots \\ \vdots \\ \vdots \\ k_{CO_2} \end{pmatrix} \Rightarrow E = \left(Y^{-1} \Delta RF_\alpha^{Global} \right) \odot K_{CO_2}^{-1} \tag{S14}$$

where the decay profile $y_{CO2}(t)$ can be transformed into square matrix $\boldsymbol{Y}$, k_{CO2} becomes a single column vector $\boldsymbol{K_{CO2}}$ with its value in every row, and through rearrangement we can now solve for $\boldsymbol{E}$ which takes the form of a vector containing annual CO_2-equivalent emissions from year $t_0 \rightarrow t_n^1$. The discrete annual albedo CO_2-eq. emission profile, E, is re-inserted into Eq. (S12) to ensure that it does indeed replicate $\Delta RF_a^{Global}(t)$.

Snow Albedo Uncertainty

Some studies examining the influence of albedo changes on radiative forcing following land surface change have chosen to utilize "snow-free" surface albedo measurements and perform a separate snow parameterization, noting that uncertainty in albedo measurements can stem largely from the influence of snow [11-14]. However, the accuracy of the MOD43 snow albedo product has been validated with ground-based albedo observations from automatic weather stations spanning 16 field locations over spatially homogeneous snow and semi-homogeneous ice-covered surfaces on the Greenland ice sheet [15]; when considering only the highest quality results from the BRDF algorithm, the MODIS albedo root mean square error (RMSE) was ± 0.04, which was only ± 0.005 higher in uncertainty than the in-situ measurements. These validation efforts have shown that the MOD43 algorithm – which includes shadowing models – captures the proper geometric-optical effects over both flat, uniform, pure snow surfaces and non-uniform or winter canopy-laden surfaces [16]. We therefore feel confident in the quality of the actual-sky albedo data adopted for use in this study, which is based only the highest quality [17] cloud-cleared results obtained from the MODIS BRDF/albedo algorithm.

---

[1] $\odot$ denotes cell by cell multiplication

Radiative Forcing Model Uncertainty

We test the reliability of the Fu-Liou radiative transfer model by benchmarking the change in mean annual forcing when annual albedo is increased by a value reported in [11] for a region with similar solar insolation and climate (pixel highlighted in yellow, Figure S3 below). A 0.001 increase in annual albedo results in a mean annual forcing of -0.1 W/m2, aligning well with [11].
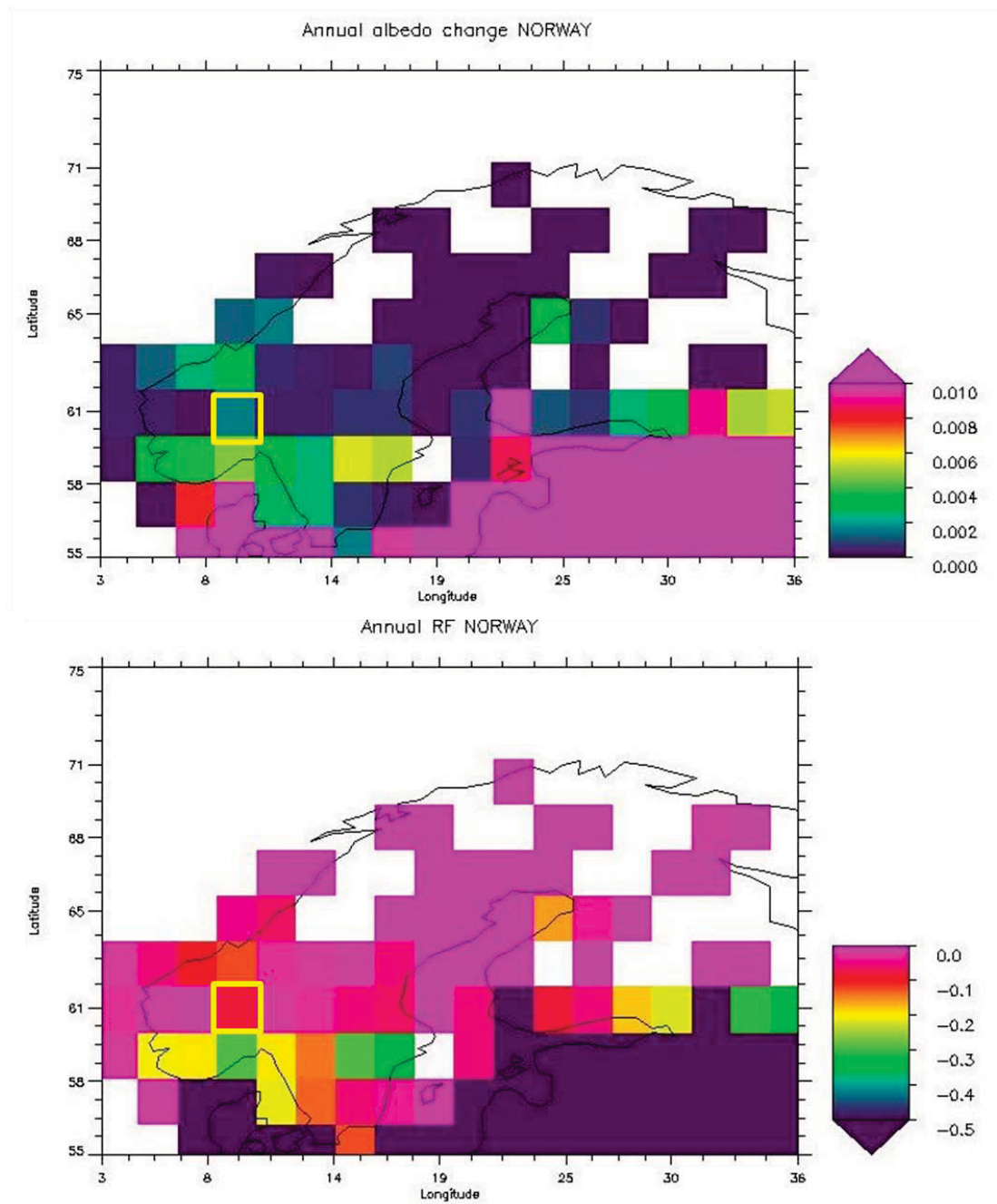
Figure SX. Annual albedo change and corresponding forcing for Norway, provided by [18]. For study details, we refer the reader to the original study [11].

Logging Scenarios

Logging scenarios are based on defining species-specific gross annual stemwood outtake volumes, presented in Table S3. These volumes remain constant for each year in both scenarios. Forestry information on site quality as a function of age and species is used to define outtake scenarios also as a function of age, changing at every 10-yr. intervals. The *percentage* of any species i of age a to the total volume of i harvested in any given time step is equal in both scenarios. Table S3 provides information on the gross annual outtake volume by species for the two scenarios.

Table S3. Total annual stem wood outtake volumes, under bark ("u.b."). Residue outtake as a percentage of all branches, bark, and foliage (i.e, "slash") generated at final felling.

Scenario	*BAU*	*BF*
	2011-2111	2011-2111
Spruce (x 10^6 m^3, u.b./year)	5.8	10
Pine (x 10^6 m^3, u.b./year)	1.8	3
Birch (x 10^6 m^3, u.b./year)	0.6	2
Residues (% slash/year)	10%	50%

We assume that 90% of the harvesting in Norway is due to clear-cut methods (as opposed to selective harvesting). The increase in outtake volume associated with the figures presented in Table S3 for the *BF* scenario results in a ~90% increase in the amount of area that is annually clear-cut.

Annual Biogeochemical Emission Fluxes from Land and Fuel Use

Figure S4 presents annual biogenic carbon fluxes in productive forests over the analysis period, disaggregated by carbon pool, for both the *FR* and *BF* scenarios. Because we are

dealing with an emission scenario, carbon "gains" (G) due to net sequestration in living biomass and soil associated with net primary production (carbon input flux) are expressed here as removals and are negative, which then requires adding "losses" (carbon output flux). In the context of IPCC guidelines, however, the expression for accounting for net carbon fluxes in forests, ΔC, is equal to gains minus losses, where carbon accumulation is positive. In our case, we are interested in looking at the flux changes between our two scenarios at any given time step, thus a negative ΔG value represents an "additional" [19, 20] net removal flux resulting from anthropogenic activity leading to enhanced sink capacity. Shown in Table S4, ΔG turns negative in the fifth decade following the switch to the BF scenario due to the effects of a shifting age distribution, whereby younger forests begin to remove carbon at higher rates.

At any time step, the net change in the biogenic carbon flux summed over all carbon pools when going from the FR to BF scenario can be expressed as:

$$\Delta C(t) = \sum_{CPools} \left(C_{Gains}^{BF}(t) - C_{Gains}^{FR}(t) \right) + \sum_{CPools} \left(C_{Losses}^{BF}(t) - C_{Losses}^{FR}(t) \right) \tag{S15}$$

or:

$$\Delta C(t) = \Delta C_{Gain}(t) + \Delta C_{Loss}(t) \tag{S16}$$

Since the "ΔC_{Loss}" flux is attributed entirely to biofuel in our BF scenario, the full amount of "ΔC_{Gain}" may be viewed as an emission credit/debit attributed to the biofuel sector.
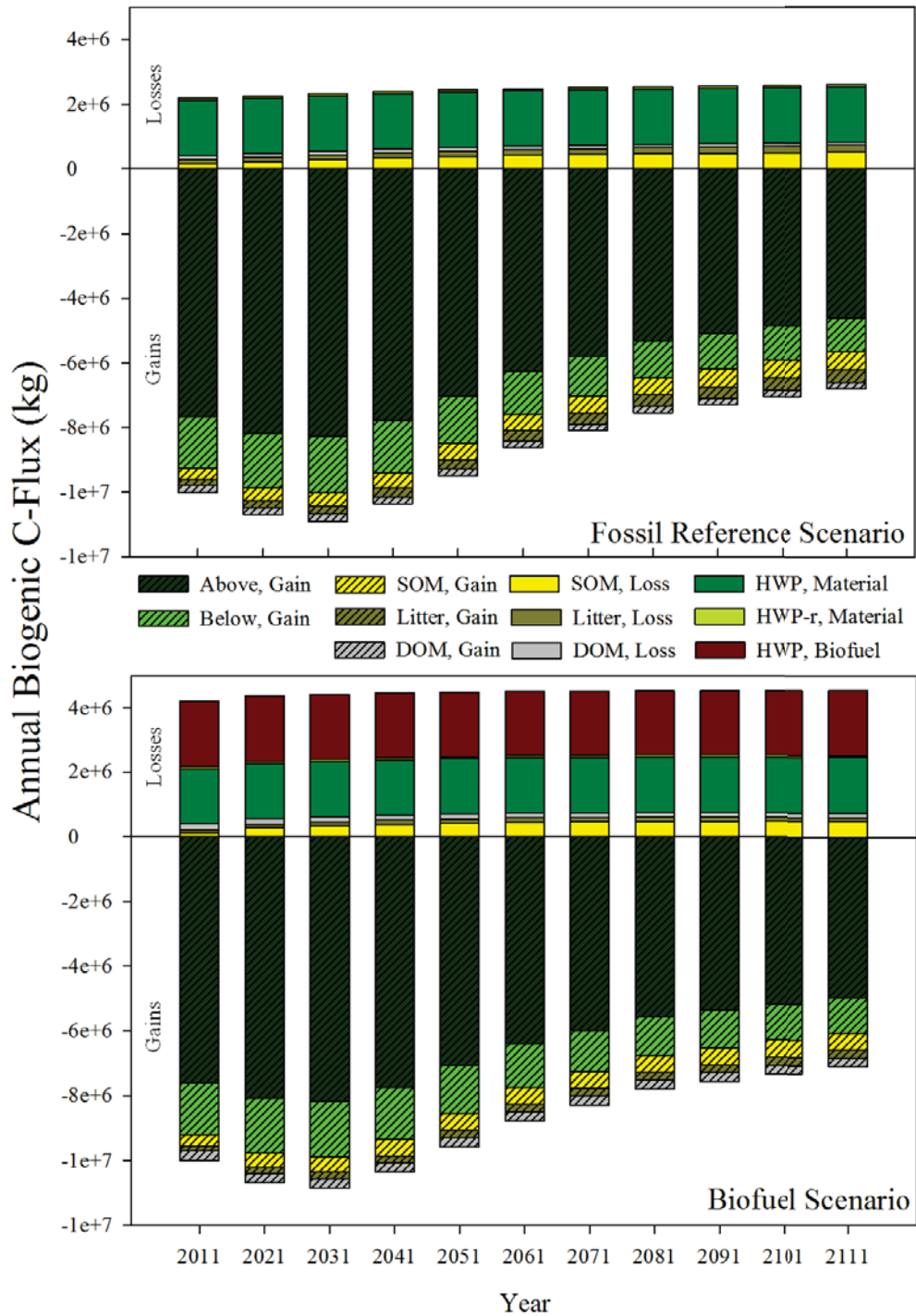
Figure S4. Annual biogenic carbon flux on productive forest areas (kg-C/year) for the *Fossil Reference* and *Biofuel* scenarios, disaggregated by IPCC pool: "*HWP, Material*" = Harvested

wood products as material; "*HWP-r, Material*" = Harvested wood product residues as material; "*HWP, Biofuel*" = HWP + HWP-r as biofuel; "*SOM*" = Soil organic matter; "*DOM*" = Dead organic material; "*Above*" = Above ground biomass; "*Below*" = Below ground biomass; "*Litter*" = Litter. The "*HWP, Biofuel*" flux of the *BF* scenario is the direct biogenic C-emission flux from biofuel production and use.

The annual aggregated biogenic carbon flux converted to CO_2, as well as life cycle fossil-fuel emissions associated with transport fuel production and use, including those associated with forestry operations, are presented in Table S5. Relative to the *FR* scenario, we see that forest management in the *BF* scenario leads to both increases and decreases in removals, $\Delta Gain$, (i.e, $Gain^{BF} - Gain^{FR}$) over a 100-year period.

Table S5. Annual biogenic carbon fluxes (Mt-CO_2/year), FTD[2] and diesel fuel use (tera-joule/year, "TJ/yr"), and the associated well-to-wheel ("WTW") fossil-based emissions of the *FR* and *BF* scenarios (tonne/TJ).

	2011	2021	2031	2041	2051	2061	2071	2081	2091	2101	2111
FR, Biogenic emission fluxes on productive forest areas											
Gain (Mt-CO_2/year)	-36.7	-39.3	-40.0	-38.0	-34.9	-31.6	-29.7	-27.7	-26.8	-25.8	-24.9
Loss (Mt-CO_2/year)	8.0	8.3	8.5	8.8	9.0	9.1	9.2	9.3	9.4	9.5	9.6
BF, Biogenic emission fluxes on productive forest areas and from biofuel production and consumption											
Gain (Mt-CO_2/year)	-36.7	-39.2	-39.8	-38.0	-35.1	-32.2	-30.4	-28.6	-27.8	-26.8	-26.0
Loss (Mt-CO_2/year)	15.5	16.1	16.3	16.5	16.6	16.6	16.6	16.7	16.7	16.7	16.7
BF-FR, Net C-flux, Land Use											
ΔGain (Mt-CO_2/year)	0.05	0.06	0.18	0.01	-0.26	-0.57	-0.73	-0.92	-0.95	-1.00	-1.04

---

[2] Calculated FTD fuel production is based on a biomass-to-FTD conversion efficiency of 45% (Lower Heating Value (LHV) basis) and a weighted-average heating value for all species and their constituents of 19.8 TJ/tonne-dry matter (LHV). Age- and species-specific biomass stem volume is converted to mass using biomass expansion factors obtained from ref 21.   Lehtonen, A., et al., *Biomass expansion factors (BEFs) for Scots pine, Norway spruce, and birch according to stand age for boreal forests.* Forest Ecology and Management, 2004. **188**: p. 211-224.

ΔLoss (Mt-CO_2/year)	7.46	7.81	7.76	7.68	7.59	7.51	7.43	7.34	7.26	7.18	7.10
Net (ΔG + ΔL)	7.51	7.87	7.94	7.69	7.33	6.94	6.70	6.42	6.31	6.18	6.06
Fuel Production & Use, Both Scenarios											
TJ/year	38,599	38,599	38,586	38,485	38,311	38,155	38,008	37,869	37,744	37,636	37,535
FR, WTW Emissions											
t-CO_2/TJ	83	83	83	83	83	83	83	83	83	83	83
t-CH_4/ TJ	4.2E-3	4.2E-3	4.2E-3	4.2E-3	4.2E-3	4.2E-3	4.2E-3	4.2E-3	4.2E-3	4.2E-3	4.2E-3
t-N_2O/ TJ	2.5E-6	2.5E-6	2.5E-6	2.5E-6	2.5E-6	2.5E-6	2.5E-6	2.5E-6	2.5E-6	2.5E-6	2.5E-6
BF, WTW Emissions (Fossil Only)											
t-CO_2/ TJ	17	17	17	17	17	17	17	17	17	17	17
t-CH_4/ TJ	8.4E-4	8.4E-4	8.4E-4	8.4E-4	8.4E-4	8.4E-4	8.4E-4	8.4E-4	8.4E-4	8.4E-4	8.4E-4
t-N_2O/ TJ	5.1E-7	5.1E-7	5.1E-7	5.1E-7	5.1E-7	5.1E-7	5.1E-7	5.1E-7	5.1E-7	5.1E-7	5.1E-7

References

1. Forster, P., et al., *Changes in atmospheric consituents and in radiative forcing*, in *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, S. Soloman, et al., Editors. 2007, Cambridge University Press: Cambridge, U. K. and New York, NY, USA.
2. Myhre, G., et al., *New estimtes of raditive forcing due to well mixed greenhouse gases.* Geophysical Research Letters, 1998. **25**(14): p. 2715-2718.
3. Joos, F., et al., *Global warming feedbacks on terrestrial carbon uptake under the Intergovernmental Panel on Climate Change (IPCC) emission scenarios.* Global Biogeochemical Cycles, 2001. **15**(4): p. 891-907.
4. Joos, F., *Pulse response functions are cost-efficient tools to model the link between carbon emissions, atmospheric CO2, and global warming.* Physics and Chemistry of the Earth, 1996. **21**: p. 471-476.
5. Oak Ridge National Laboratory Distributed Active Archive Center (ORNL DAAC), *MODIS subsetted land products, Collection 5. Available online [http://daac.ornl.gov/MODIS/modis.html] from ORNL DAAC, Oak Ridge, Tennessee, USA. Accessed 30 September, 2010.* 2010, ORNL DAAC.
6. Fu-Liou, *Fu-Liou online 200507 (Diurnal Simulation). http://snowdog.larc.nasa.gov/cgi-bin/rose/flp200503/sun/flsun.cgi .* 2005.
7. ISCCP. *International Satellite Cloud Climatology Project. http://isccp.giss.nasa.gov/*.
8. Norwegian Forest and Landscape Institute, *Forest Cover in Norway*, in *Accessed January 6, 2011 at: www.miljostatus.no:* Ås.
9. Nilson, T. and U. Peterson, *Age Dependence of Forest Reflectance: Analysis of Main Driving Factors.* Remote Sensing of Environment, 1994. **48**: p. 319-331.
10. Nilson, T., et al., *Forest Reflectance Modeling: Theoretical Aspects and Applications.* Ambio, 2003. **32**(8): p. 535-541.

11. Myhre, G., M.M. Kvalevåg, and C.B. Schaaf, *Radiative forcing due to anthropogenic vegetation change based on MODIS surface albedo data.* Geophysical Research Letters, 2005. **32**: p. L21410.

12. Betts, R.A., *Offset of the potential carbon sink from boreal forestation by decreases in surface albedo.* Nature, 2000. **408**: p. 187-190.

13. Bird, D.N., et al., *Incorporating changes in albedo in estimating the climate mitigation benefits of land use change projects.* Biogeosciences Discuss., 2008. **5**: p. 1511-1543.

14. Myhre, G. and A. Myhre, *Uncertainties in Radiative Forcing due to Surface Albedo Changes Cause by Land-Use Changes.* Journal of Climate, 2003. **16**: p. 1511-1524.

15. Stroeve, J., et al., *Accuracy assessment of the MODIS 16-day albedo product for snow: comparisons with Greenland in situ measurements.* Remote Sensing of Environment, 2005. **94**: p. 46-60.

16. Moody, E.G., et al., *Northern Hemisphere five-year average (2000-2004) spectral albedos of surfaces in the presence of snow: Statistics computed from Terra MODIS land products.* Remote Sensing of Environment, 2007. **111**(2-3): p. 337-345.

17. *"High quality" in the context of the MODIS albedo product has a very specific meaning and refers to retrievals having seven or more non-obscured observations with sufficient angular sampling over the product's 16-day period to provide a well-defined BRDF.*

18. Kvalevåg, M.M., *Changes in Surface Albedo and Radiative Forcing in Norway*. 2010, Pesonal communication, November 10, 2010.

19. Searchinger, T.D., *Biofuels and the need for additional carbon.* Environmental Research Letters, 2010. **5**(2): p. 10.

20. Searchinger, T.D., et al., *Fixing a Critical Climate Accounting Error.* Science, 2009. **326**(5952): p. 527-528.

21. Lehtonen, A., et al., *Biomass expansion factors (BEFs) for Scots pine, Norway spruce, and birch according to stand age for boreal forests.* Forest Ecology and Management, 2004. **188**: p. 211-224.