SCIENCE • TECHNOLOGY • RESEARCH HIGHLIGHTS • VISIONS

Dissertation

11

Assessing the uncertainties of climate policies and mitigation measures

Viewpoints on biofuel production, grid electricity consumption and differentiation of emission reduction commitments

Sampo Soimakallio

VTT SCIENCE 11

Assessing the uncertainties of climate policies and mitigation measures

Viewpoints on biofuel production, grid electricity consumption and differentiation of emission reduction commitments

Sampo Soimakallio

Thesis for the degree of Doctor of Science (Tech.) to be presented with due permission of the School of Engineering for public examination and criticism in Auditorium K216, at the Aalto University School of Engineering (Espoo, Finland) on the 31st of August 2012 at 12 noon.

ISBN 978-951-38-7469-8 (soft back ed.)
ISSN 2242-119X (soft back ed.)

ISBN 978-951-38-7470-4 (URL: http://www.vtt.fi/publications/index.jsp)
ISSN 2242-1203 (URL: http://www.vtt.fi/publications/index.jsp)

JULKAISIJA – UTGIVARE – PUBLISHER

VTT
PL 1000 (Tekniikantie 4 A, Espoo)
02044 VTT
Puh. 020 722 111, faksi 020 722 7001

VTT
PB 1000 (Teknikvägen 4 A, Esbo)
FI-02044 VTT
Tfn. +358 20 722 111, telefax +358 20 722 7001

VTT Technical Research Centre of Finland
P.O. Box 1000 (Tekniikantie 4 A, Espoo)
FI-02044 VTT, Finland
Tel. +358 20 722 111, fax + 358 20 722 7001

Kopijyvä Oy, Kuopio 2012

Assessing the uncertainties of climate policies and mitigation measures
Viewpoints on biofuel production, grid electricity consumption and differentiation of emission reduction commitments

Ilmastopolitiikkatoimien ja päästövähennysten epävarmuuksien arviointi. Näkemyksiä biopolttoaineiden tuotannosta, verkkosähkön kulutuksesta ja päästövähennysvelvoitteiden taakanjaosta. **Sampo Soimakallio.** Espoo 2012. VTT Science 11. 78 p. + app. 80 p.

Abstract

Ambitious climate change mitigation requires the implementation of effective and equitable climate policy and GHG emission reduction measures. The objective of this study was to explore the significance of the uncertainties related to GHG emission reduction measures and policies by providing viewpoints on biofuels production, grid electricity consumption and differentiation of emission reduction commitments between countries and country groups. Life cycle assessment (LCA) and macro-level scenario analysis through top-down and bottom-up modelling and cost-effectiveness analysis (CEA) were used as methods. The uncertainties were propagated in a statistical way through parameter variation, scenario analysis and stochastic modelling.

This study showed that, in determining GHG emissions at product or process level, there are significant uncertainties due to parameters such as nitrous oxide emissions from soil, soil carbon changes and emissions from electricity production; and due to methodological choices related to the spatial and temporal system boundary setting and selection of allocation methods. Furthermore, the uncertainties due to modelling may be of central importance. For example, when accounting for biomass-based carbon emissions to and sequestration from the atmosphere, consideration of the temporal dimension is critical. The outcomes in differentiation of GHG emission reduction commitments between countries and country groups are critically influenced by the quality of data and criteria applied. In both LCA and effort sharing, the major issues are equitable attribution of emissions and emission allowances on the one hand and capturing consequences of measures and policies on the other. As LCA and system level top-down and bottom-up modelling results are increasingly used to justify various decisions by different stakeholders such as policy-makers and consumers, harmonization of practices, transparency and the handling of uncertainties related to methodological choices, parameters and modelling must be improved in order to avoid conscious misuse and unintentional misunderstanding.

Keywords greenhouse gas emission, biofuel, electricity, effort sharing, uncertainty

Ilmastopolitiikkatoimien ja päästövähennysten epävarmuuksien arviointi
Näkemyksiä biopolttoaineiden tuotannosta, verkkosähkön kulutuksesta ja päästövähennysvelvoitteiden taakanjaosta

[Assessing the uncertainties of climate policies and mitigation measures – Viewpoints on biofuel production, grid electricity consumption and differentiation of emission reduction commitments]. **Sampo Soimakallio.** Espoo 2012. VTT Science 11. 78 s. + liitt. 80 s.

Tiivistelmä

Kunnianhimoiset tavoitteet ilmastonmuutoksen hillitsemiseksi edellyttävät tehokkaiden ja oikeudenmukaisten ilmastopolitiikka- ja päästövähennystoimenpiteiden toteuttamista. Tämän tutkimuksen tavoitteena oli analysoida kasvihuonekaasupäästöjen vähentämiseen liittyvien keinojen ja politiikkatoimenpiteiden epävarmuuksia tarkastelemalla biopolttoaineiden tuotantoa ja verkkosähkön kulutusta sekä päästövähennysvelvoitteiden taakanjakoa maiden ja maaryhmien välillä. Menetelminä käytettiin elinkaariarviointia, makrotaloustason skenaarioanalyysia ja kustannustehokkuusanalyysia. Epävarmuuksia tarkasteltiin tilastollisten menetelmien avulla mm. parametrien oletuksia vaihtelemalla, skenaarioanalyysilla ja stokastisella mallintamisella.

Tulokset osoittavat, että tuote- tai prosessitasolla biopolttoaineiden tuotannon ja verkkosähkön kulutuksen kasvihuonekaasupäästöihin liittyy merkittäviä epävarmuuksia, joita aiheutuu arvioinnissa käytettävistä parametrioletuksista, esimerkiksi maaperän typpioksiduulipäästöille ja hiilivaraston muutoksille sekä sähköntuotannon päästöille. Epävarmuuksia aiheutuu myös tarkastelujen rajauksista ja allokointikäytännöistä sekä mallinnukseen liittyvistä tekijöistä, kuten biomassan hiilen vapautumisen ja sitoutumisen välisen ajallisen esiintymisen käsittelemisestä. Maa- tai maaryhmätasolla päästövähennysvelvoitteiden taakanjaossa sovellettavat kriteerit ja tietopohja ovat kriittisiä tulosten kannalta. Sekä elinkaariarvioinnissa että taakanjaossa päästöjen ja päästövähennysvelvoitteiden oikeudenmukainen kohdentaminen ja kerrannaisvaikutusten arvioiminen ovat keskeisiä tekijöitä ja voivat edellyttää useiden erilaisten menetelmien käyttämistä. Elinkaariarvioinnin ja järjestelmätason mallinnuksen tuloksia käytetään enenevässä määrin erilaisten päätösten perusteena. Tarkoitushakuisen väärinkäytön ja tarkoituksettomien väärinymmärrysten välttämiseksi on erittäin tärkeää, että elinkaariarviointiin ja järjestelmätason mallinnukseen liittyviä käytäntöjä yhtenäistetään, tulosten ja oletusten läpinäkyvyyttä lisätään ja menetelmiin, parametreihin ja mallinnukseen liittyvien epävarmuuksien käsittelyä parannetaan.

Avainsanat greenhouse gas emission, biofuel, electricity, effort sharing, uncertainty

Preface

It was the spring of 2000, when I started my career as a scientist at VTT Technical Research Centre of Finland. In the early stages my work was mainly related to a techno-economic assessment of abatement of fluorinated greenhouse gases. Since then I have participated in many projects regarding greenhouse gas impacts related to energy systems, in particular bioenergy systems. I have also become familiar with various measures for differentiating emission reduction targets between nations. The ideas and papers in this thesis have been generated over many years and result from several public research projects. I kindly thank all the funding organizations involved (see acknowledgement for details). The processing of the two latest papers and this summary was funded by VTT, and my gratitude goes especially to Vice President Kai Sipilä and Technology Manager Seppo Hänninen.

I express my greatest gratitude to the Climate Change Research Group of VTT. I thank especially Professor Ilkka Savolainen, who acted as an instructor of this thesis, for his support, motivation and guidance throughout the last 12 years. I also wish to thank Professor Sanna Syri for her support, encouragement and guidance. She is a former member of the Group and my former line manager, and she acted as a supervisor of this thesis. Furthermore, I would like to highlight the role of Dr Kim Pingoud and Mikko Hongisto, who have both taught me a great deal regarding methodologies for determining greenhouse gas emissions in various contexts during these years. I am also grateful to all the other – former and current – Group members for a vast number of fruitful conversations and collaboration related to climate change mitigation in its all comprehensiveness.

I would like to thank the pre-examiners of my thesis, Professor Kornelis Blok and Professor Anders Hammer Strømman for their dedication and valuable comments on my thesis. Naturally, the co-authors of my thesis played a significant role. Every one of them have taught me something special. Consequently, I would like to express my warmest gratitude to; Tommi Ekholm, Juha Kiviluoma, Kati Koponen, Tuula Mäkinen, Teuvo Paappanen (who unexpectedly passed away in 2011) and Professor Ilkka Savolainen (VTT), Dr. Laura Saikku and Dr. Hannu Mikkola (University of Helsinki), Dr. Katri Pahkala (MTT Agrifood Research Finland), Professor Sanna Syri (Aalto University) and Dr. Niklas Höhne and Sara Moltmann (Ecofys GmbH).

I would like to acknowledge numerous other colleagues in particular at VTT, Finnish Environment Institute SYKE, MTT Agrifood Research Finland, Finnish Forest Research Institute (Metla), The Government Institute for Economic Research (VATT), University of Helsinki, Statistics Finland, the Finnish Ministry of the Environment, the Ministry of Employment and the Economy and IEA Bioenergy Task 38 network for fruitful collaboration and many interesting and pleasant discussions throughout these years. Also, the people at the Energy Systems knowledge centre of VTT not least for the in many ways useful and enjoyable coffee breaks are gratefully acknowledged. For valuable assistance in drawing Figure 2 of the thesis, I would like to thank Kati Koponen and Sébastian Piquemal.

Furthermore, I would like to thank all my dear friends outside my work. The many amusing things that have happened, and the unique time which we have spent together, especially those of my friends who are involved in Hissien Ystävät ry, are something totally indispensable. Tennis and floorball have been my dearest hobbies for a long time. I am really happy to have had an opportunity to get to know many very nice people off and on the tennis courts for almost 30 years. The same holds true of floorball, in which the Hanat Auki ry team with all the former and current members have played an important role in my life for more than 15 years.

Finally, I would like to thank all my dear family and relatives, especially my wife Laura, parents Ulla and Anssi, brother Mikko (and family) and my parents-in-law Marjatta and Olli. In particular, my parents Ulla and Anssi have supported me unstintingly all these years. Despite their endless love and support, I think they did not believe that I would ever do this. Never mind, neither did I. But then I got to know Laura. It was autumn 2007 when we met for the first time in the Finnish Ministry of the Environment. (In the meanwhile I would like to thank retired Senior Environmental Adviser Jaakko Ojala for unintentionally bringing us together). Laura encouraged me to publish some of my work in scientific journals. She helped me a great deal in getting to know how papers should be produced. Later I became totally inspired with writing papers and became more confident that I would write this thesis someday. Two very special and important places can be highlighted; the dear winter holiday house Monomaja in Äkäslompolo and our own home in Kruununhaka. Both of them have played an invaluably important role in the creation and writing process of the last three papers and this summary. Besides being my dear and loving wife and the joy of my life, Laura really inspired and encouraged me to write this thesis. It is impossible to express here how grateful I am to her.

Espoo, June 2012
Sampo Soimakallio

Academic dissertation

Supervisor	Professor Sanna Syri Energy Economics Department of Energy Technology Aalto University, School of Engineering, Finland
Instructor	Professor Ilkka Savolainen VTT Technical Research Centre of Finland
Reviewers	Professor Kornelis Blok Copernicus Institute Utrecht University, the Netherlands
	Professor Anders Hammer Strømman Program for Industrial Ecology Faculty of Engineering Science and Technology Department of Energy and Process Engineering Norwegian University of Science and Technology, NTNU Trondheim, Norway
Opponent	Professor Markku Ollikainen Department of Economics and Management University of Helsinki

List of papers

This thesis is based on the following original publications, which are referred to in the text as I–VI (Appendix B).The publications are reproduced with kind permission from the publishers.

I **Soimakallio, S.**, Mäkinen, T., Ekholm, T., Pahkala, K., Mikkola, H., Paappanen, T. 2009. Greenhouse gas balances of transportation biofuels, electricity and heat generation in Finland – Dealing with the uncertainties. Energy Policy 37(1), 80–90.

II **Soimakallio, S.** & Koponen, K. 2011. How to ensure greenhouse gas emission reductions by increasing the use of biofuels? – Suitability of the European Union sustainability criteria. Biomass & Bioenergy 35, 3504–3513.

III **Soimakallio, S.**, Kiviluoma, J., Saikku, L. 2011. The complexity and challenges of determining GHG (greenhouse gas) emissions from grid electricity consumption and conservation in LCA (life cycle assessment) – A methodological review. Energy 36, 6705–6713.

IV **Soimakallio, S.**, Saikku, L. 2012. CO_2 emissions attributed to annual average electricity consumption in OECD (the Organisation for Economic Co-operation and Development) countries. Energy 38(1), 13–20.

V Saikku, L**.** & **Soimakallio, S.** 2008. Top-down approaches for sharing GHG emission reductions: uncertainties and sensitivities in the 27 European Union Member States. Environmental Science & Policy 11(8), 723–734.

VI Ekholm, T., **Soimakallio, S.**, Moltmann, S., Höhne, N., Syri, S. & Savolainen, I. 2010. Effort sharing in ambitious, global climate change mitigation scenarios. Energy Policy 38(4), 1797–1810.

Author's contributions

In **Paper I**, Sampo Soimakallio is the main author. Sampo Soimakallio structured the paper, selected the methodological choices, collected the greenhouse gas emission and uncertainty data and created the assessment model in MS Excel software. Tuula Mäkinen planned the research questions, provided data on the ethanol process and commented on the paper. Tommi Ekholm carried out the Monte Carlo simulations. Density functions for parameters were jointly selected by Tommi Ekholm and Sampo Soimakallio. Hannu Mikkola and Teuvo Paappanen provided data on the energy use of agricultural and forest machinery, respectively. Katri Pahkala provided data on agrochemical use and agricultural yield rates.

In **Paper II**, Sampo Soimakallio is the main author. The analysis was jointly planned by Sampo Soimakallio and Kati Koponen. The paper was structured and written by Sampo Soimakallio. Kati Koponen collected the data for the literature research, wrote the supplementary information and commented on the paper.

In **Paper III**, Sampo Soimakallio is the main author. The analysis was planned by Sampo Soimakallio and Juha Kiviluoma. The paper was structured and written by Sampo Soimakallio. Juha Kiviluoma and Laura Saikku commented on the paper.

In **Paper IV**, Sampo Soimakallio is the main author. The paper was co-written by Sampo Soimakallio and Laura Saikku. The calculations were carried out by Sampo Soimakallio. The GHG emission intensity analysis was carried out by Sampo Soimakallio and the country level emission leakage analysis by Laura Saikku. The data was jointly collected by Sampo Soimakallio and Laura Saikku.

In **Paper V**, the data was jointly collected and analysed and the manuscript co-written by Laura Saikku and Sampo Soimakallio. Laura Saikku was responsible of the scenario setting and Sampo Soimakallio conducted the sensitivity analysis.

In **Paper VI**, Tommi Ekholm is the main author. The analysis was jointly planned by Tommi Ekholm, Sampo Soimakallio and Niklas Höhne. Triptych and Multistage calculations were carried out by Sara Moltmann. ETSAP-TIAM calculations were carried out by Tommi Ekholm. The paper was designed and written by Tommi Ekholm, while Sampo Soimakallio, Sara Moltmann, Niklas Höhne, Sanna Syri and Ilkka Savolainen participated in the project and commented on the paper.

Contents

List of symbols and abbreviations

ALCA	Attributional life cycle assessment
BTL	Biomass-to-Liquid
CB	consumption-based
CBA	Cost-benefit analysis
CCS	carbon capture and storage
CEA	Cost-effectiveness analysis
CLCA	Consequential life cycle assessment
CH_4	methane
CHP	combined heat and power production
CO_2	carbon dioxide
CO2-eq.	carbon dioxide equivalent
DFE	design for environment
dLUC	direct land-use change
DM	dry material
DME	dimethylether
EIT	Economies in Transition
EJ	eksajoule (10^{18} joules)
EtOH	ethanol
ETS	Emission Trading Scheme
ETSAP	Energy Technology Systems Analysis Programme
EUROSTAT	the statistical office of the European Union
FAME	fatty acid methyl ester

FT	Fischer-Tropsch
EVOC	Evolution of Commitments
GE	general equilibrium
GDP	gross domestic product
GHG	greenhouse gas
GWP	Global Warming Potential
HVO	hydrotreated vegetable oil
IEA	International Energy Agency
iLUC	indirect land-use change
IMAGE	the Integrated Model to Assess the Global Environment
IOA	input-output analysis
IPCC	the Intergovernmental Panel on Climate Change
LCA	life cycle assessment
LCC	life cycle costing
LCI	life cycle inventory
LCIA	life cycle impact assessment
LHV	lower heating value
LUC	land-use change
LULUCF	Land use, land-use change and forestry
MAC	marginal abatement curve
MCDA	multiple criteria decision analysis
MFA	material flow analysis
MS	Member State (of the European Union)
N_2O	nitrous oxide
non-ETS	all sectors outside the European Union Emission Trading Scheme
OECD	the Organisation for Economic Co-operation and Development
PB	production-based
PE	partial equilibrium
ppm	parts per million
PPP	Purchasing power parity

RED	Renewable Energy Directive
RF	radiative forcing
RME	rapeseed methyl ester
SEEA	Systems for economic and environmental accounts
SETAC	the Society for Environmental Toxicology and Chemistry
SLCA	social life cycle assessment
SRES	Special Reports on Emission Scenarios
TIAM	TIMES Integrated Assessment Model
TIMES	The Integrated MARKAL (market allocation) – EFOM (Energy Flow Optimisation Model) System
UNEP	The United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change

1. Introduction

1.1 Background

The Earth's environment has been unusually stable for the past 10,000 years (Dansgaard et al. 1993, Petit et al. 1999, Rioual et al. 2001). This stability may now be under threat due to human actions which have become the main driver of global environmental change. This change has been most intensive since the industrial revolution and in particular since the Second World War. Three of nine[1] interlinked planetary boundaries – for a safe operating space for humanity – climate change, rate of biodiversity loss and interference with the nitrogen cycle have already been overstepped (Rockström et al. 2009). In addition, the boundaries for global freshwater use, change in land use, ocean acidification and interference with the global phosphorous cycle may soon be approached. Furthermore, various boundaries are tightly coupled. If one boundary is exceeded, then the others are also under serious risk.

Climate change is the major, primarily environmental issue of our time, and the single greatest challenge facing environmental regulators (UNEP 2012). There is a large scientific consensus that increasing atmospheric concentrations of greenhouse gases have an increasing impact on the global mean surface temperature (IPCC 2007a). The increase in the global temperature may have serious and irreversible impacts on the ecosystems, leading to increasing crisis also for human systems as regards for instance food production, health and safety, and economy. The extent, strength and timing of the implications are not well-known, but are very likely more serious the more the global mean surface temperature increases (IPCC 2007b).

Climate change results from the altered energy balance of the climate system and is driven by changes in the atmospheric concentrations of greenhouse gases (GHGs) and aerosols, changes in land cover and in solar radiation (IPCC 2007a). The positive or negative changes in energy balance due to these factors are ex-

[1] Climate change; rate of biodiversity loss (terrestrial and marine); interference with the nitrogen and phosphorous cycles; stratospheric ozone depletion; ocean acidification; global freshwater use; change in land-use; chemical pollution; atmospheric aerosol loading.

pressed as radiative forcing, which is used to compare warming or cooling influences on global climate (ibid.). Increased atmospheric concentrations of GHGs result in positive radiative forcing tending raise the temperature, whereas anthropogenic contributions to aerosols, surface albedo through land-use changes and depletion of stratospheric ozone produce a cooling effect (ibid.). There is a very high confidence that the global average net effect of human activities since 1750 until 2005 has been one of warming, with a radiative forcing of +1.6 W/m^2 with an uncertainty range from +0.6 to +2.4 W/m^2 (ibid.). Carbon dioxide contributes most significantly to radiative forcing. In 2011 the annual mean atmospheric concentration of carbon dioxide equalled approximately 392 ppm with an annual growth rate of around 2 ppm (NOAA 2012).

Measured as carbon dioxide equivalents based on global warming potential (GWP) over 100 years, the contribution of CO_2 emissions from fossil fuel combustion was approximately 57% of all anthropogenic GHG emissions in 2004 (IPCC 2007c). Similarly, the corresponding contribution of CO_2 emissions from deforestation and decay of biomass was 17%, CO_2 emissions from other sources 3%, methane emissions 14%, N_2O emissions 8% and hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulphurhexafluoride (SF_6) together 1% (ibid.). Some 26% of the GHG emissions resulted from energy supply, 20% from industry, 17% from forestry, 14% from agriculture, 13% from transport, 8% from residential and commercial buildings and 3% from waste and wastewater (ibid.).

Anthropogenic GHG emissions have increased significantly since pre-industrial times (IPCC 2007c). CO_2 emissions from fossil fuel combustion in particular have increased rapidly during recent decades (Peters et al. 2012). Only economic recessions, namely the oil crisis (1973), the US savings and loan crisis (1979), the collapse of the former Soviet Union (1989), the Asian financial crisis (1997) and the global financial crisis (2008–2009) have temporarily reduced the annual level of these emissions. In 2010, the combined CO_2 emissions from fossil fuel combustion and cement production were the highest ever, equalling 33.4±1.8 Gt CO_2 (Peters et al. 2012). The major emitting countries or country groups in absolute terms were China (26%), USA (17%), EU (12%), India (7%), Russia (5%) and Japan (4%). CO_2 emissions from deforestation and biomass decay have been remarkable in some countries, especially in Indonesia and Brazil in recent decades, thus increasing the contribution of those countries to overall GHG emissions (Houghton 2009).

Countries also contribute to emissions through globalization. The emissions embodied in traded products are becoming increasingly important (Peters et al. 2012, Davis & Caldeira 2010, Peters et al. 2009, Peters & Hertwich 2008). Of the global carbon dioxide emissions in 2008, 26% (7.8 Gt CO_2) were shifted around the globe due to international trade (Peters et al. 2011). Developing countries often produce goods that are used in developed countries. Net fossil CO_2 emission transfers from developing to developed countries increased fourfold between 1990 and 2010.

1.2 Climate policy

The ultimate objective of the United Nations Framework Convention on Climate Change (UNFCCC 1992) is the stabilisation of atmospheric concentrations of GHGs at a level that prevents dangerous anthropogenic interference with the climate system. Furthermore, “such a level should be achieved within a time frame sufficient to allow ecosystems to adapt naturally to climate change, to ensure that food production is not threatened and to enable economic development to proceed in a sustainable manner”. Article 3.1 of the UNFCCC requires that the mitigation effort should be shared between the parties “on the basis of equity and in accordance with their common but differentiated responsibilities and respective capabilities”. However, the UNFCCC does not determine any concrete requirements. Thus, among others acceptable limit for global mean surface temperature growth, the emission target levels of various countries and the timing of emission reductions were left open. The Kyoto Protocol (1997) under the UNFCCC obligated industrialised countries to reduce their GHG emissions by 5.2% from the 1990 level on the average between 2008 and 2012. The USA did not ratify the Protocol. The European Union (EU) and later all the countries that have ratified the UNFCCC (1992) have recognized "the scientific view that the increase in global temperature should be below 2°C (EC 1996, 2007, UNFCCC 2010). The Conference of the Parties of the UNFCCC held in Durban in 2011 agreed to reach a new comprehensive climate protocol, another legal instrument or agreed outcome with legal force concerning all the parties (UNFCCC 2011a). However, many details, including the exact form of the agreement and the interpretation of its legal validity as well as emission reduction commitments, remained still open.

According to the IPCC, global GHG emissions should peak no later than 2015 and be reduced by at least 50–85% by 2050 and perhaps even more than 100% prior to the end of the century from their levels in 2000 in order to retain a reasonable probability of limiting the global mean surface temperature increase to under 2°C compared to the pre-industrial level (IPCC 2007c). However, the uncertainties involved in climate modelling are significant. One important but little known parameter is climate sensitivity to increasing atmospheric concentrations of GHGs (IPCC 2007a). In addition, most climate models do not consider long-term reinforcing feedback mechanisms that may further warm the climate, such as decreasing ice cover. Consequently, recently used climate models may underestimate the impacts of increasing atmospheric GHG concentrations (Hansen et al. 2008).

Halting global mean surface temperature increase would require significant improvement in the level of ambition of GHG emission reductions by the parties (UNFCCC 2011a). In order to reach a global solution in climate negotiations, the equity issue has to be solved. Any effort-sharing principle should be politically acceptable with respect to fairness principles and operational requirements (Torvanger & Ringius 2001). The key issue with an effort-sharing method is the dilemma between its transparency, on the one hand, and its ability to take into account national circumstances, on the other (Soimakallio et al. 2006). Each country has to have the impression that it is treated equitably relative to the others in order for it to participate. By the end of 2011, the viewpoints of the major emitters concerning binding emission reduction targets and effort sharing between countries, have been too diverse for a breakthrough in climate negotiations under the UNFCCC.

The EU has unilaterally committed itself to reduce its GHG emissions by at least 20% from the 1990 level by 2020 (EC 2008). Within the EU, GHG emissions are regulated under the two levels: the EU Emission Trading Scheme (ETS) including mainly GHG emissions from energy production and industry (EU 2009a), and at a national level including sectors such as residential, agriculture, transportation and waste management not incorporated in the EU ETS (EU 2009b). As a part of the integrated climate and energy package, the EU also introduced mandatory targets to increase by 2020 the share of renewable energy sources in final energy consumption to 20% and in transportation to 10% (EU 2009c). As a part of this Renewable Energy Directive (RED), mandatory sustainability criteria were introduced for transportation biofuels and other bioliquids, to be accounted for in the targets and allowed to benefit from subsidies. The EU has also set itself a target by 2020 of reducing its primary energy consumption by 20% compared to projections (EC 2011a). In the long term, the EU is committed to reduce GHG emissions by 80–95% by 2050 from their 1990 level in the context of necessary reductions by developed countries as a group (EC 2011b). To achieve its long-term target, the EU has published, among others, roadmaps for resource efficiency (EC 2011c) and energy (EC 2011d).

1.3 GHG emission reduction measures

Ambitious climate change mitigation will require effective climate policy and GHG emission reductions in all countries and all sectors. The emission reduction measures related to energy production and use include improved energy efficiency of the economies, reduced deforestation, fuel switching from coal to gas and from fossil fuels to biofuels (solid, gaseous or liquid), nuclear power, wind power, hydro power, solar and geothermal energy and carbon dioxide capture and storage (CCS). Other options include improved agricultural practices, afforestation, reforestation, forest management, harvested wood product management, recycling and waste and wastewater management. The cost effectiveness and reduction potential of different emission reduction measures vary significantly across countries and sectors. According to van Vuuren et al. (2009), the largest reduction potential as a response to carbon prices exists in the energy supply sector, whereas emission reductions in the building sector may carry relatively low costs. According to IEA (2010a), improvement of end-use fuel and electricity efficiency provides the greatest potential for a substantial reduction in energy-related CO_2 emissions. According to IPCC (2007c), most of the least cost potential for technical emission reduction measures in 2030 exists in non-OECD/EIT countries and in buildings, agriculture, forestry and energy supply.

The uncertainties related to actual GHG emissions in addition to technical, economic and ecological issues, as well as externalities and the development of costs result in uncertainty in the cost-efficiency and potential of use of various emission reduction measures. For example, the forecasted long-term overall availability of bioenergy varies significantly between studies, from some 40–80 EJ/a to over 1,000 EJ/a in the most pessimistic and optimistic scenarios respectively (Bringezu et al. 2009). Expert review of the IPCC (2011) concluded that the potential could be in the range of 100 to 300 EJ/a by 2050. A number of recent studies have concluded that the increased production of biofuels may cause significant environmental and social problems, and that GHG emission reductions achieved by substituting fossil fuels with biofuels, especially liquid biofuels, are unclear due to the auxiliary material and energy inputs required, the direct land-use impact and, in particular, indirect impacts such as deforestation (Searchinger et al. 2008, Fargione et al. 2008, Righelato & Spraclen 2007, Plevin et al. 2010, Runge et al. 2007, Reijnders & Huijbregts 2008, Mitchell 2008, Doornbosch & Steenblik 2007, de Santi et al. 2008, Edwards et al. 2010, Soimakallio et al. 2009). Uncertainty about the interaction of the energy sector with the rest of the economy in its turn increases the uncertainty related to the introduction of various emission reduction measures (Weyant 2000).

1.4 Aims of the study

The fundamental aim of this study is to explore the significance of the uncertainties related to GHG emission reduction measures and policies. Regarding emission reduction measures, the GHG balances of using biomass as transportation biofuels and in heat and electricity production in Finland are studied. Furthermore, the suitability of the European Union sustainability criteria for ensuring GHG emission reductions by increasing the use of transportation biofuels is analysed. In addition, the determination of GHG emissions related to grid electricity consumption at product or process level is studied in general and on average annual basis in OECD countries in particular. Regarding emission reduction policies, effort sharing in ambitious global climate change mitigation scenarios up to 2050 and in the EU by 2020 is studied. The importance of methodological choices and parameter assumptions on the results as well as equity issues are analysed and discussed. Finally, suggestions are given for the way forward.

2. Theoretical framework

The interactions between human activities and the environment can be systematically analysed through industrial ecology (Socolow et al. 1994). The fundamental aims of industrial ecology are to close the loop of materials and substances, and to reduce resource consumption as well as environmental impacts. It is a descriptive discipline, and furthermore a normative discipline, as many industrial ecologists are concerned about the potential environmental impacts of production and consumption, and trying to ascertain how things ought to be, and finding ways to achieve the goals (Lifset & Graedel 2002). Industrial ecology overlaps with many other research fields such as engineering, ecological economics and environmental management. It is neither purely scientific nor purely technological, but includes elements of both.

In industrial ecology several tools from product level to global analysis are utilised. The family of material flow analysis (MFA) are basic analytical tools for industrial ecology derived from the first law of thermodynamics: energy cannot be created or lost (den Hond 2000, Bringezu et al. 1997). At the product or process level, lifecycle assessment (LCA) extends to these analyses by attempting to quantify the environmental impacts of the use of materials and substances, in particular product or process systems (Rebitzer et al. 2004). The resulting environmental profile of a product or process can be used for comparison with competing products or processes or for proposing ways to enhance the particular product or process design through design for environment (DFE) (den Hond 2000). At the global or regional levels, the IPAT concept[2] to study dematerialisation and the effects of technology as well as changes in population and affluence on changes in the environment is used in industrial ecology (Chertow 2000). Furthermore, systems for economic and environmental accounts (SEEA) are established and developed within many countries to be applied at regional or national level (Finnveden & Moberg 2005). In SEEA, environmental input-output analysis (IOA) is used for assessing environmental impacts from different sectors (Finnveden & Moberg 2005). Different types of policy models such as general equilibrium (GE) and partial equilibrium (PE) models are also widely used to provide scenario data at global or regional level.

[2] Environmental impact (I) = Population (P) x Affluence (A) x Technology (T).

3. Material and methods

3.1 Methodological framework

3.1.1 Product level analysis

Life Cycle Assessment (LCA) is a methodological framework for estimating and assessing the environmental impacts related to the life cycle of a product system (product, process or service) (ISO 2006:14040, 2006:14044). Two main categories of LCA have been defined: attributional and consequential (Finnveden et al. 2009, Curran et al. 2005). The Attributional LCA (ALCA) has been defined as a method "to describe the environmentally relevant physical flows of a past, current, or potential future product system". In contrast, the Consequential LCA (CLCA) can be defined as a method that aims to describe how environmentally relevant physical flows would have been or would be changed in response to possible decisions that would have been or would be made. The ALCA reflects the system as it is, whereas the CLCA attempts to respond to the question: "What if?". The Attributional LCA excludes the use of marginal data. Instead, some sort of average data reflecting the actual physical flows is used (Finnveden et al. 2009). In contrast, in Consequential LCA marginal data is used when relevant for the purpose of assessing the consequences (Ekvall and Weidema 2004).

The LCA is initiated by defining the goal and scope; this is followed by a life cycle inventory (LCI), a life cycle impact assessment (LCIA) and an interpretation of the results (ISO 2006:14040). Definition of the appropriate system boundary and other methodological choices, for example allocation methods and functional unit, depend on the goal and scope of the study. Reflecting the iterative nature of LCA, decisions regarding the data to be included should be based on a sensitivity analysis to determine their significance (ISO 2006:14044). Allocation is one of the major unsolved issues in LCA. According to ISO standards, it should be avoided whenever possible by dividing the unit process to be allocated into two or more sub-processes or by expanding the product system to include all the additional functions related to co-products. If allocation cannot be avoided, it should reflect the underlying physical relationships between products or functions or be based on other relationship. (ISO 2006:14044.)

Uncertainty is involved in every step of LCA from the goal and scope definition to interpretation. According to Huijbregts (2001), the uncertainty in LCA is due to 1) methodological choices such as spatial and temporal system boundary, functional unit and allocation procedure, 2) parameters such as inaccurate or outdated measurements or lack of data, and 3) models such as loss of spatial and temporal dimension when accounting for emissions and derivation and application of characterization factors. In addition, variation in the results is due to spatial and temporal variability and variability in objects and sources. The ISO 14040 and 14044 does not give concrete guidance on how the uncertainties should be analysed. According to Finnveden et al. 2009, uncertainty can be handled in several ways. The "scientific" way to deal with large uncertainties is to do more research to lower the uncertainty; the "social" way is to discuss the uncertain issues with stakeholders and to find a consensus. The "statistical" way does not try to remove or reduce the uncertainty, but intends to incorporate it. For the latter option, a number of methods are available, including parameter variation and scenario analysis, classical statistical theory on the basis of probability distributions, tests of hypothesis, Monte Carlo simulations and other sampling approaches, analytical methods based on first-order error propagation, non-parametric analysis, Bayesian analysis, Fuzzy set theory and qualitative uncertainty methods (Finnveden et al. 2009).

In this study LCA is applied to assess GHG emissions of transportation biofuels and biomass-based power and heat production in Finland by considering the reference fuels to be substituted (Paper I). Transportation biofuel technologies for which GHG emissions were not previously studied in Finland were selected for consideration. Critical issues resulting in uncertainty of the LCA are considered in the "statistical" way. The significance of parameter uncertainty is reflected for the technologies considered. Previously, only a few LCA studies have conducted parameter uncertainty analysis by using stochastic simulation (Williams et al. 2009, Lloyd & Ries 2007).

The importance of setting a system boundary and selecting allocation methods is studied for determining CO_2 emissions from annual average electricity consumption in OECD countries (Paper IV). Previous studies have examined the GHG emissions of single electricity production technologies (Weisser 2007), the impact of allocation method on CO_2 emissions from CHP (e.g. Graus & Worrel 2011, Frischknecht 2000) and the uncertainty of CO_2 emission intensities at various geographic levels in the continental US (Weber et al. 2010). Also, the role of international trade on GHG emissions in general has been studied (e.g. Peters & Hertwich 2008). However, the above-mentioned issues have not been studied comprehensively and transparently together to a wider extent for a range of countries.

Furthermore, the significance and suitability of selection between the ALCA and CLCA approach, the setting of spatial and temporal system boundary, the selection of allocation methods and sources of parameter uncertainty are critically discussed in the context of grid electricity consumption in general (Paper III) and in the context of the sustainability criteria for transportation biofuels and other bioliquids introduced as a part of Renewable Energy Directive (RED) of the EU (Paper II). Regarding electricity consumption, only a few studies overall have been published previously on the methodological issues and data uncertainties, and a comprehensive picture was lacking. In addition, the suitability of the mandatory sustainability criteria to ensure the GHG emission reductions by increasing the use of transportation biofuels and other bioliquids was analysed and discussed critically for the first time in Paper II.

3.1.2 Global and regional level analysis

Macro-level scenarios describing the relations between the economy, the energy sector and the environment can be carried out by using two different modelling approaches called top-down and bottom-up (IPCC 2007c). Top-down modelling describes the macro-economic relations in the region under consideration, thus evaluating the system through aggregate economic variables. Top-down models may apply rather simple descriptions of, for example, country-level future development of energy consumption by primary energy sources and economic sectors. On the contrary, bottom-up modelling includes detailed descriptions of all the processes involved. In order in bottom-up models to construct a scenario, the development of all the parameters needs to be specified, and the impacts of individual factors or interlinkages of various factors are considered.

In this study, effort sharing of emission reduction commitments between countries and country-groups are analysed by applying both top-down and bottom-up modelling. The uncertainty is propagated in the "statistical" way. A few top-down approaches based on macro-economic figures are studied for sharing the national emission reduction targets between the EU Member States by 2020 (Paper V). The effort sharing at a global level up to 2050 was studied based on two top-down approaches, namely Triptych and Multistage (Höhne et al. 2006) and analysed by using the bottom-up partial equilibrium energy system model ETSAP-TIAM (Loulou & Labriet 2008, Loulou 2008, Syri et al. 2008, Koljonen et al. 2009) under different socio-economic baseline scenarios (Paper VI). ETSAP-TIAM model has not previously been used to analyse the emission reduction and cost implications of effort sharing. Cost-effectiveness analysis (CEA)[3] was applied as a methodology to characterize the cost implications.

3.2 System description and data

Six different papers are included in this study (Table 1). In four of the papers (I–IV) LCA is applied as a methodological framework, of which two are related to GHG

[3] CEA is a form of economic analysis and a special case of cost-benefit analysis (CBA) in which all the costs of a portfolio of projects (e.g. GHG emission reduction costs) are assessed in relation to a policy goal such as a GHG emission reduction target (Sathaye et al. 1993). CBA is a systematic process to measure all the negative and positive impacts and resource uses of a project, decision or government policy in the form of monetary costs and benefits (Squire & van der Tak 1975, Ray 1984).

emissions of biofuels (I, II) and two concerning GHG emissions of grid electricity consumption (III, IV). Top-down modelling is applied in Paper V, and both top-down and bottom-up modelling are applied in Paper VI to consider GHG emission reduction effort sharing in the context of climate policy. One of the six Papers (IV) is retrospective in nature and concerns only CO_2 emissions, whereas the others are future-oriented covering all the relevant GHG emissions. In characterizing GHG emissions, two of the Papers (I, V) clearly rely on Global Warming Potentials calculated by using 100-year time frame (GWP-100). In Paper VI both Radiative Forcing (RF) and GWP-100 factors are applied. Sections 3.2.1–3.2.6 below provide an overview of the system considered and the major data sources used in each of the papers. More detailed description is provided in the respective papers.

Table 1. Illustrative description of the scope and type of the papers.

	Paper I	**Paper II**	**Paper III**	**Paper IV**	**Paper V**	**Paper VI**
Characterization	Data-oriented	Discussion	Discussion	Data-oriented	Data-oriented	Data-oriented
Technology/ sector	Biomass-based transportation fuels, electricity, heat	Transportation biofuels and other bioliquids	Electricity	Electricity	Non-ETS sector	All sectors excl. LULUCF
Region	Finland	EU-27	not specified	OECD	EU-27 MSs	Global in 15 regions
Time	Future-oriented, not specified	2020	Future-oriented, not specified	1990–2008	2020	2020, 2050
Emission components	GHGs	GHGs	GHGs	CO_2	GHGs	GHGs
Emission characterization	GWP-100 (IPCC 1996)	GWP-100 (IPCC 2001) / not specified	Not specified	Not considered	GWP-100 (IPCC 1996)	GWP-100 (IPCC 1996) / RF
Methodological framework	CLCA	ALCA/CLCA	ALCA/CLCA	ALCA	Sectoral top-down	Sectoral top-down / bottom-up, CEA
Main type of uncertainty considered	Parameter	Methodological choices, Parameter, model	Methodological choices	Methodological choices	Methodological choices, Parameter	Methodological choices, Parameter
Methods for uncertainty propagation	Stochastic modelling	Not considered	Not considered	Parameter and system boundary variation	Parameter variation and scenario analysis	Parameter variation and scenario analysis

3.2.1 GHG balances of biofuels in Finland (Paper I)

In this paper, GHG emission reductions of biomass used as transportation fuels, and in heat and electricity production in Finland when replacing reference fuels are assessed. Principles of the CLCA approach were followed. Allocation was avoided through system expansion. One kilometre driven and one kilowatt hour produced were selected as functional units for transportation fuels and electricity/heat production respectively. Parameter uncertainty analysis was conducted by using Monte Carlo simulation with 15,000 samples. Calculations were carried out using MS Excel software (vs 2003) and its add-in software Crystal Ball (vs 2000). The transportation biofuel technologies considered include ethanol from barley, rape methyl ester (RME) diesel from (spring) turnip rape, Fischer Tropsch (FT) diesel from logging residues and reed canary grass.

For FT diesel production, three different process concepts were assumed, including stand alone process and processes integrated into a pulp and paper mill which minimizes either electricity or biomass consumption. In addition, electricity and/or heat production from logging residues and reed canary grass were considered. Fossil diesel was considered as a reference fuel for RME and FT diesel, and gasoline was considered as a reference fuel for ethanol. Marginal electricity with its assumed minimum and maximum values was considered to provide boundaries for calculating the credits of replacing electricity and/or heat production by biofuels.

It was assumed that no commercial reference use for the raw materials takes place. Agrobiomass-based raw materials were assumed to be cultivated on set-aside lands, whereas logging residues were assumed to be left in the terrain in the reference situation. Protein animal meal generated in the ethanol and RME biodiesel process was assumed to replace the use of soy protein imported from the USA. Glycerine produced in RME process was assumed to be used for energy in heat production boilers to replace peat. Straw was not assumed to be harvested.

Unit processes considered include auxiliary energy inputs (crude oil, diesel oil, electricity), auxiliary chemical inputs (fertilizers, limestone, pesticides, sulphuric and phosphoric acid, smectite, caustic soda and hexane) and soil processes (N_2O emissions from fertilization, CO_2 emissions from limestone and changes in soil carbon balances). The construction of infrastructure, the production of facilities, machinery and other equipment required in overall fuel production chains were excluded from both bioenergy and reference fuel chains.

Data on cultivation, harvesting, transportation and crushing of biomass raw materials was based on Mäkinen et al. 2006. Intensities for direct and indirect N_2O emissions from soils due to fertilization were derived from IPCC (2006) and Statistics Finland (2006). Data on compensation fertilization of forest lands and soil carbon losses due to logging residue harvesting was based on Wihersaari (2005). Data on soil carbon changes due to agricultural land management was taken from IPCC (2006). Data on biofuel processing chemicals and energy balance of RME diesel processing was taken from Elsayed et al. (2003). Data on processing of the other fuels and combustion of the fuels was based on Mäkinen et al. 2006. Data on the supply of diesel oil, heavy fuel oil and natural gas required in machinery and equipment, pesticides and substitution credits of soybean meal was based on Edwards et al. (2003). CH_4 and N_2O emissions from fuel combustion in machinery and boilers and specific fuel consumption and the GHG emissions of transport were derived from Statistics Finland (2006) and LIPASTO calculation system of VTT (2006). Data on substitution credits from peat combustion was derived from Kirkinen et al. (2007).

All variables were presented with a three-parameter Weibull distribution and determined as uncorrelated. An exception to this was GHG emissions from electricity consumption and substitution, for which a uniform distribution was assigned. The uncertainty range given for each variable was based on the data sources used and expert evaluation.

3.2.2 EU sustainability criteria analysis (Paper II)

According to the sustainability criteria introduced in the Renewable Energy Directive (RED) of the EU, the GHG emission reductions compared to fossil comparator should be at least 35% for biofuels and other bioliquids produced before the end of 2016. From the beginning of 2017, the GHG emission reductions should be at least 50% and from the beginning of 2018, the GHG emission saving should be at least 60% for biofuel production installations where production begins after 1 January 2017.

The RED provides the default values for GHG emission reductions (%) of a range of biofuels compared to fossil reference fuels. These default values can be used if GHG emissions from land-use changes can be proved to be equal to or less than zero. In addition, the RED provides disaggregated default values, separately and as aggregate, for cultivation, fuel processing and transport and distribution for a range of biofuels expressed as g CO_2-eq./MJ_{fuel}. Disaggregated default values for cultivation can only be used if the raw materials are cultivated outside the European Community, are cultivated in the Community areas included in the specific list referred to in the RED, or are waste or residues from other than agriculture, aquaculture and fisheries. If the above mentioned conditions are not fulfilled, if the default value for the GHG emission saving from a specific production pathway falls below the required minimum level or if the default value does not exist, biofuel producers are required to use the RED methodology to show that the actual GHG emission reductions resulting from their production process fulfil the set criteria. Furthermore, the biofuel producer may always use the actual value instead of the default value.

The GHG emission reduction is defined as the relative reduction compared to reference fuel by the Equation:

$$\text{EMISSION SAVING} = (EF - EB) / EF \tag{1}$$

where,

E_B = total emissions from the biofuel or other bioliquid; and
E_F = total emissions from the fossil fuel comparator.

Equation 1 takes into account the GHG emissions from the different phases from cultivation (crops) or collection (waste and residues) of raw materials to the use of biofuel. GHG emissions from the production of machinery and infrastructure are excluded. Allocation should be based on lower heating value of the products in the case of co-products other than electricity. The other details of the formula are given in the part C of Annex V of the RED. For the implementation of the RED into national legislation of the EU Member States, the European Commission issued two Communications. These include practical guidelines on the implementation of the sustainability system and the associated calculation rules (EC 2010a), and a Communication on voluntary certification systems and default values (EC 2010b). In addition, a Decision on the calculation of land carbon stocks in the case of land-use changes was issued (EC 2010c).

In Paper II, the conservativeness of the default values provided in the RED for GHG emission reductions (%) compared to fossil reference fuels for a range of biofuels was analysed by comparing them to figures presented in the literature. In addition, the methodology introduced in the RED to calculate actual GHG emission reductions was analysed considering the most critical issues, problems and challenges that are encountered when assessing life cycle GHG emissions of transportation biofuels and other bioliquids in general.

3.2.3 Determination of GHG emissions of electricity consumption (Paper III)

Electricity cannot be stored as such, and is therefore consumed virtually at the same time as it is produced. Electricity can, however, be transmitted over even long distances via overhead lines and power cables. Within an electrical network, the consumption and thus also the production typically varies between times of day, seasons and years. Furthermore, the electricity production mix varies from one moment to another, and can be very different in different electrical grids. These specific properties make the assessment of GHG emissions associated with the individual process of consuming or conserving grid electricity a complex and challenging procedure. However, the particular information is highly relevant and required for almost any environmental impact assessment in one form or another.

In Paper III, a methodological review of the complexity and challenges of determining GHG emissions from individual processes that consume or conserve grid electricity was carried out by means of a literature survey. The critical issues and uncertainties involved were discussed. The viewpoints of ALCA and CLCA approaches were reflected.

3.2.4 CO_2 emission intensity of electricity in OECD countries (Paper IV)

In Paper IV, the CO_2 emission intensity of annual average electricity consumption in the 30 OECD countries was examined in 1990, 1995 and 2000–2008 by both ignoring and considering the CO_2 emissions embodied in the electricity trade. First, the annual production-based CO_2 emission intensity of electricity (g CO_2/kWh) was calculated by determining the total CO_2 emissions from fuel combustion in power production and dividing this by the total amount of electricity produced and transferred to consumption points within a country. In the production-based approach, it was assumed that electricity imports to a country have the same CO_2 emission intensity as the electricity produced within the particular country.

Second, the CO_2 emissions embodied in the electricity trade were calculated and the consumption-based CO_2 emission intensity of electricity (g CO_2/kWh) was estimated. In the case where an OECD country imports electricity from a non-OECD country, the production-based CO_2 emission intensity of electricity supply for the non-OECD country in question was calculated. In cases where the origin of electricity import was not known or no reliable data was available (electricity imports from Luxembourg to Germany between 1990 and 2000), the production-based CO_2 emission intensity of the OECD average was applied.

Two different methods were selected for allocation of CO_2 emissions from combined heat and power production (CHP) to heat and power. For the lower limit of CO_2 emissions attributed to electricity, emissions were allocated on an equal basis to electricity and heat output in enthalpic terms. For the upper limit of power-related CO_2 emissions from CHP, the 'motivation electricity' method was selected, allocating 100% of the emissions to electricity.

The latest available data from the International Energy Agency (IEA) was used. The CO_2 emissions from fuel combustion, categorised as electricity output from the main electricity producers, autoproducers and combined heat and power producers, as well as own use of electricity, were taken from the IEA database 'CO_2 emissions from fuel combustion' (IEA 2010b). The data for electricity production, distribution and transformation losses, imports, exports and final consumption, as well as electricity and heat production in CHP plants was taken from the IEA database 'Energy Balances' (IEA 2010c). The data for bilateral electricity trade of the OECD countries was taken from the IEA publication 'Electricity Information' (IEA 2010d). The overall national CO_2 emission data was taken from the UNFCCC (2011b).

3.2.5 Effort sharing in the EU by 2020 (Paper V)

In Paper V, top-down macro-level figures were used to set the emission reduction targets for the 27 Member States of the EU. Four effort-sharing criteria were generated for emission reduction in sectors outside the Emission Trading Scheme (ETS) referred as non-ETS. In Scenario 1, the annual rate of change in GHG/GDP was assumed to be the same in all Member States over the 13 years 2008–2020. In Scenario 2 it was assumed that GHG/GDP converges for all countries by 2020. In Scenario 3 it was assumed that national annual rates of GHG/GDP development are the same as they were in 1993–2005. In order to reach a reduction of 20% by 2020, an additional reduction was required. This additional annual reduction was set as a constant over time and the same for all countries in percentage terms. In Scenario 4 it was assumed that per capita GHG emissions converge for all countries by 2020. The reduction in the non-ETS sector was determined through reductions in the ETS sector. In the ETS sector, each country was hypothetically set to reduce its emissions by the same proportion compared to their verified ETS sector emissions in 2005. The first year when emission reduction requirements were assumed to take place was 2008.

A few test runs were conducted for all scenarios to analyze certain sensitivities involved in the results. In the test runs, the base year (starting point for reductions) for emissions and GDP was changed. In addition, the period for ETS reductions was changed from the latest verified emissions to allocated future emissions. In addition, ETS reductions as a proportion of the total reduction were changed. Moreover, GDP and population forecasts were varied.

The historical data for GHG emissions and GDP, as well as forecasts for population growth in the different EU Member States was derived from the Eurostat database (Eurostat 2008). Forecasts of economic development were carried out according to a model described in Saikku et al. (2008). GDP estimates for the non-ETS sectors were used in the calculation. The approximated GDP share of the sectors included in the ETS was based on Eurostat (2008) GDP data. Required GHG emission intensities were compared to recent historical development in the scenarios. Historical developments in GHG/GDP during 1993–2005 were calculated for total GDP. Non-ETS GHG estimates for 1993 were based on Eurostat (2008). GDP data for 1993 were taken from the Penn World Table (Heston et al. 2007).

3.2.6 Global effort sharing up to 2050 (Paper VI)

Paper VI focuses on the equity of effort sharing with two exogenously assumed reduction targets that would stabilize greenhouse gas atmospheric concentrations to 485 ppm CO_2-eq. and 550 ppm CO_2-eq. by year 2100. The corresponding GHG emission developments from 1990 were +20% (by 2020) and -50% (by 2050) and +30% (2020) and -10% (2050), respectively. The emission level of 2050 was assumed to be constant for the period between 2050 and 2100. Based on assumptions on global emission paths, the resulting atmospheric GHG concentrations, radiative forcing and global mean surface temperature increase (using 3°C climate sensitivity) up to 2100 were calculated.

A relatively simple and transparent tool, Evolution of Commitments (EVOC), was used to calculate the effort sharing based on Triptych and Multistage approaches (Höhne et al. 2006). Such allocations of emissions were then analysed in long-term energy-climate scenarios produced with ETSAP-TIAM (Loulou & Labriet 2008, Loulou 2008, Syri et al. 2008, Koljonen et al. 2009), a more sophisticated integrated assessment model.

The EVOC tool contains collections of data on emissions from several sources, and future projections of relevant variables from the Integrated Model to Assess the Global Environment (IMAGE) implementation of the IPCC SRES scenarios marked as A1, A2, B1 and B2. As emission data varies in its completeness and sectoral split, EVOC combines data from the selected sources and harmonizes it with respect to the sectoral split. Future emissions are based on IMAGE projections of parameters, such as population, GDP (PPP), electricity consumption and industrial value added. As IMAGE projections are available only for 17 world regions, EVOC de-aggregates this data by combining it with historical values. Finally, the user can set the parameters of several effort sharing rules in order to calculate emission allocations.

In the Triptych approach (Phylipsen et al. 1998, Groenenberg et al. 2001, den Elzen et al. 2008a) the emission target for each sector is calculated with given assumptions on the reduction potentials in the sector. The Triptych version 6.0 that was used in the study is documented by Phylipsen et al. (2004). This version uses six sectors: Electricity, Industry, Fossil fuel production, Domestic, Agriculture and Waste. The electricity and industry sectors use parameters on efficiency, structure and income levels to calculate the emission limits. Domestic and waste sectors use a single convergence level, given in terms of t CO_2-eq./capita, to which the emissions of countries converge by a given year. For fossil fuel production and agriculture, reduction levels from the baseline are assumed. In addition to this sectoral differentiation, Triptych also uses a rough income categorization with some parameters to distinguish countries with different levels of affluence. The emission allocation of a country is then the sum of the sectoral targets.

In the Multistage approach the countries participate in several stages with differentiated levels of commitment (den Elzen et al. 2006). Each stage has stage-specific commitments with countries graduating to higher stages when they exceed certain thresholds, and all countries agree to have commitments at a later point in time. For this study, thresholds and commitments were applied based on per capita emissions with four stages. The cap-and-trade system was assumed to bind all countries so that the countries without binding commitments receive emission allocations according to their baseline emissions, but are then free to mitigate emissions and sell the excess allowances for profit.

The energy and emission scenarios in this paper were devised using the ETSAP-TIAM (TIMES Integrated Assessment Model) which is based on the TIMES (The Integrated MARKAL-EFOM System) modelling methodology (Loulou et al. 2005). The TIMES family of models are bottom-up type linear partial equilibrium models that calculate the market equilibrium through the maximization of the total discounted economic surplus with given external end-use demand projections. The ETSAP-TIAM models the whole global energy system with 15 geographical regions. Main assumptions concerning the energy system, future energy technologies, potentials, other emission reduction options and climate module in the model are described in Syri et al. (2008). All GHGs regulated under the Kyoto Protocol were considered from all anthropogenic sources, except emissions from land-use changes.

The geographical region split of the ETSAP-TIAM model was used. The externally given energy consumption in the ETSAP-TIAM model, based on the growth of regional GDP, was harmonised to fit with the four IPCC SRES scenarios considered. The GHG emission reduction costs considered include direct costs, changes in energy trade, GHG emission allowance trade and the value of lost demand due to price elasticity. Indirect macroeconomic costs, damage costs and possible benefits from avoided climate change, relevant in cost-benefit analysis (CBA), were ignored.

4. Results

4.1 Biofuels

GHG emissions from the production and use of ethanol derived from barley and RME diesel derived from turnip rape in Finland were very likely (with 94% and 98% probability, respectively) higher compared to the fossil reference fuels (Figure 1). The wide uncertainty range and high upper limit (Figure 1) resulted mainly from a significant uncertainty in N_2O emissions from soils due to fertilisation (Table 2). Other dominant factors affecting uncertainty were yield per hectare, animal feed output and emissions from electricity production. GHG emissions from producing FT diesel were lower compared to fossil diesel, but the value depended significantly on the concept considered. If the biomass requirement was minimised, GHG emissions of FT diesel were highly dependent on emissions from production of electricity consumed in the process. If the purchased electricity requirement was minimised and replaced by more biomass, the uncertainty range was decreased significantly, and soil carbon losses due to logging residue harvesting became the most dominant factor. The probability distributions for GHG emission reductions of biofuels derived from logging residues and reed canary grass were very similar compared to each other.

The GHG emission reduction in replacing electricity and/or heat by bioenergy was highly dependent on the emission factor given for the replaced energy (referred to as emission savings from replaced electricity in Table 2). The emission factor given for electricity has the opposite impact on the results in the case of replacing marginal electricity compared to consuming electricity in the case of transportation biofuels. Consequently, the higher emission factor of electricity increases the emission reduction achievable by using logging residues or reed canary grass in electricity production and decreases the emission reduction achievable by using the particular raw materials as transportation biofuels.

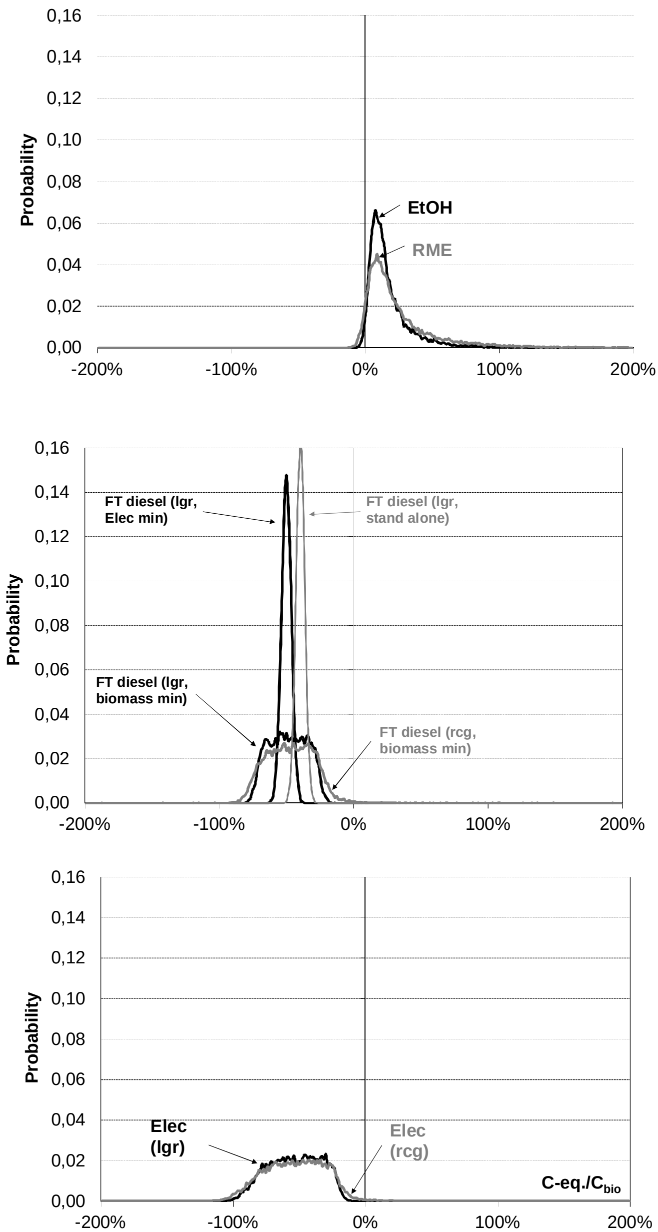


Figure 1. Probability distributions for carbon equivalent emission impact per consumed biocarbon when replacing reference fuels (Paper I). Positive values refer to emission increase. (Elec = electricity production, lgr = logging residues, rcg = reed canary grass, elec min and biomass min refer to integrated FT diesel processing cases with minimum purchased electricity and biomass, respectively.)

Table 2. Mean value, 95% central confidence interval and Spearman's rank correlation between 10 most important uncertainty variables and the GHG emission reduction per biocarbon consumed for biofuel chains studied in Paper I. (Elec = electricity production, lgr = logging residues, rcg = reed canary grass, elec min and biomass min refer to integrated FT diesel processing cases with minimum purchased electricity and biomass, respectively.) (Adapted from Paper I.)

Statistical measure	**EtOH**	**RME**	**FT (lgr, bio-mass min)**	**FT (lgr, elec min)**	**FT (lgr, stand alone)**	**FT (rcg, bio-mass min)**	**Elec (lgr)**	**Elec (rcg)**
2.5%:ile value (%)	-1%	-3%	-74%	-58%	-47%	-79%	-93%	-98%
Mean value (%)	17%	25%	-49%	-50%	-40%	-47%	-53%	-53%
97.5:ile value (%)	65%	106%	-26%	-45%	-34%	-15%	-22%	-14%
Spearman's rank correlation parameters								
Emission from electricity production	0.27	0.07	**0.97**	0.36	0.09	**0.89**		
Electricity demand			0.06	0.02	0.01	0.04		
Yield rate of raw material	-0.26	-0.27				-0.16		-0.13
Carbon content in DM of raw material	-0.07					0.15	-0.01	0.12
LHV in DM of raw material			-0.04	-0.18	-0.15	-0.04	-0.04	-0.03
N_2O from soil (fertilization)	**0.84**	**0.88**	0.04	0.14	0.01	0.25	0.03	0.20
Fertiliser use	0.12	0.09				0.03		0.02
Emissions from fertiliser production	0.10	0.11						0.02
Ploughing								-0.02
Animal feed output		0.15						
Soil carbon losses	0.16	0.14	0.21	**0.84**	**0.94**		0.13	
Emission savings from replaced electricity							**-0.95**	**-0.89**
Efficiency of biofuelled power plant							-0.27	-0.26
Emissions of biofuelled power plant							0.02	
Output of produced fuel		-0.15	-0.05	-0.21	-0.19	-0.03		
Emission savings from replaced soybean meal	-0.06	-0.06						
Emissions from replaced reference fuel	-0.05		-0.06	-0.17	-0.15	-0.05		
Emissions from transportation			0.02	0.06	0.08	0.03	0.02	0.03
Emissions from forest haulage			0.01	0.02	0.02		0.01	
Emissions from chipping			0.01	0.01	0.02		-0.01	
CO_2 from liming	0.05							
lime use		0.06						

The conservativeness of the GHG emission default values provided in the sustainability criteria of the EU Renewable Energy Directive (RED) was analysed in Paper II. Based on the literature survey, the GHG balance figures for various biofuel supply chains vary significantly around the default values provided in the RED (Figure 2). Some very high GHG emission estimates were found from the literature for biodiesel derived from palm oil and soya oil, and ethanol derived from grains. Such figures include CO_2 emissions from converting permanent forests to arable lands, directly or indirectly. Also, lower GHG emission estimates were found compared to the default values of the RED. The variation in the results for specific raw materials may be due to differences in spatial system boundary setting, handling of timing issues, allocation procedure and parameter assumptions. The 95% central confidence interval figures presented in Paper I for the relative GHG emission impact are also presented as GHG emissions of relevant biofuels[4] in Figure 2. Those figures fall in the range, with the exception that the upper limit for FT diesel from logging residues (BTL wood residues in Figure 2) was higher than any other figures found in the literature considered. On the other hand, not many figures were available for BTL from wood residues.

[4] The conversion from relative GHG emission impact has been carried out in accordance with the methodology explained in the supplementary material of Paper II (by using a GHG emission factor of 83.8 g CO_2-eq./MJ for fossil fuel replaced)

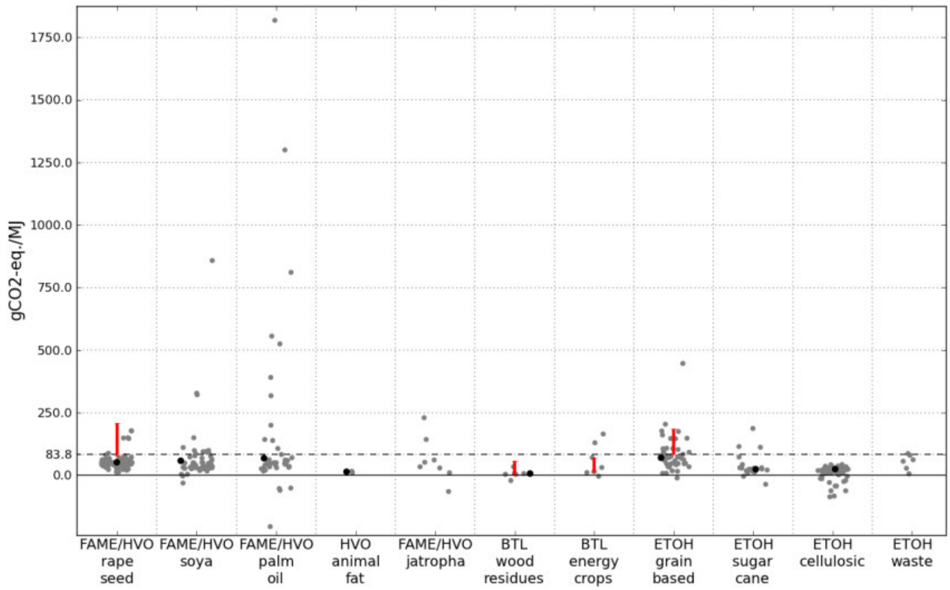


Figure 2. GHG balances of different biofuels produced from various raw materials in different regions and using different process technologies (adapted from Paper II). The black dotted line illustrates the GHG balance of the fossil reference fuel (gasoline and diesel) including CO_2 emission from fossil fuel combustion in accordance with the RED. The default values of the RED for certain raw materials and technologies are illustrated by black circles. In case the RED provides more than one default value for a certain technology route, the maximum value is presented. The vertical bars (red coloured) illustrate the range between the 95% central confidence interval of GHG emissions of biofuels studied in Paper I.

4.2 Grid electricity consumption

The variation in annual production-based CO_2 emission intensities of electricity in the countries studied in Paper IV was significantly high, ranging from almost zero in Norway during all the years studied to over 1,800 g CO_2/kWh in Poland in 1990 (Tables A1 and A2 in Appendix A). However, high values of over 1,000 g CO_2/kWh occurred only in three countries, namely Poland, the Czech Republic and Greece, during the period studied. In these countries, the use of fossil fuels, in particular coal, constituted a significant proportion of electricity production. The high values may also indicate poor quality of the original data or relatively low conversion efficiency. Apart from Norway, other examples of countries with low production-based CO_2 emission intensities were Sweden and Switzerland. The higher the fossil-fuel-based electricity production was in a given country, the higher was the CO_2 emission intensity of energy production. The share of fossil fuels in the electricity production mix varied significantly between countries (IEA 2010c).

The annual variation in production-based CO_2 emission intensity of electricity was moderate at the average OECD level, but considerable for many individual countries due to changes in the fuel mix and production technologies (Tables A1 and A2 in Appendix A). Examples of such countries are Luxembourg, Norway, Finland, Sweden, Denmark and France. For the Nordic countries, in particular, annual fluctuations in hydropower and nuclear power production significantly affected the respective amount of fuel used in electricity production.

The allocation procedure for CHP increased the variability of the results when the amount of electricity produced with CHP was high (Tables A1 and A2 in Appendix A). Examples of countries with a relatively high share of CHP in electricity production are Poland, Denmark, Finland and Sweden. Relatively, the largest range in estimated production-based CO_2 emission intensity of electricity due to the allocation procedure for CHP was in Sweden, where the lower end (energy-based allocation) CO_2 emissions totalled only 30% of the CO_2 emissions at the higher end (all for electricity) on average between 2000 and 2008. Other countries where the respective ratio due to variation was significant were Switzerland (54%), Denmark (55%), Norway (57%), and Finland (65%).

The difference between national production-based (Tables A1 and A2 in Appendix A) and consumption-based (Tables A3 and A4 in Appendix A) CO_2 emission intensity of electricity was highly significant for Switzerland, Norway, Slovakia, Austria and Sweden, and fairly significant for Denmark, Finland, Hungary and Italy (Figure 3). Of these countries, only Denmark was a net exporter of CO_2 emissions embodied in electricity trade (Figure 2 in Paper IV). For the rest of the countries studied, the difference was typically less than 10% within the years studied. The Netherlands, for example, imports a significant share of its final electricity consumption, but mainly from Germany, in which the CO_2 emission intensity of electricity production is relatively close to that of the Netherlands. For a few European countries with a high share of electricity trade compared to final electricity consumption, the CO_2 emissions embodied in electricity trade were significant compared to overall national CO_2 emissions. Such countries include Switzerland, Slovakia, Luxembourg, Austria and Finland.

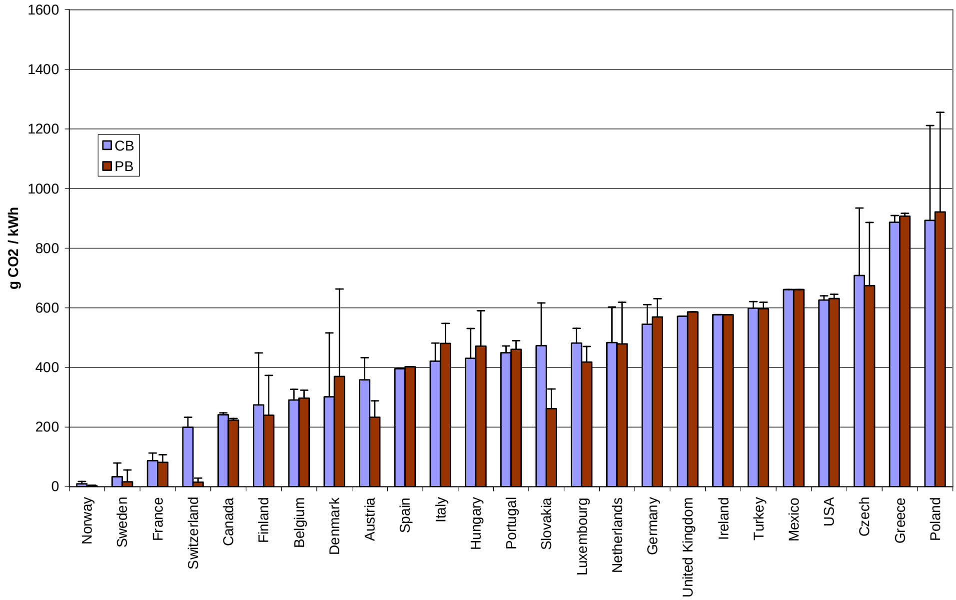


Figure 3. Production-based (PB) and consumption-based (CB) CO_2 emission intensities of electricity (g CO_2/kWh) in OECD countries with electricity trade averaged between 2006 and 2008 (Paper IV). The error bars illustrate the impact of the selected method for the allocation of CO_2 emissions between electricity and heat in combined heat and power production (CHP). The coloured columns correspond to the energy-based allocation and the upper limit of the error bars correspond to the 'motivation electricity' method.

4.3 Differentiation of emission reduction commitments

4.3.1 At the EU level by 2020

The macro-level perspective in sharing national GHG emission reduction commitments between the EU Member States was examined in Paper V with respect to achieving the 20% reduction in 1990 level GHG emissions within the European Union by 2020. Only the sectors outside the EU ETS (i.e. non-ETS), such as transportation, housing, services and agriculture, were considered.

Countries' GHG emission reduction targets were determined by their level of GHG emissions in the starting year (2008), their recent GDP and population level and growth expectations. Also, historical development in GHG/GDP had an impact in one scenario. The overall variation among the Member States in the required GHG emission reduction targets was found to be large, although the variation between scenarios was moderate for a few large EU countries (Figure 4). The required country-specific reductions were dependent on the applied principle of effort sharing, the allocation of reductions between ETS and non-ETS sectors, the selected base year for GDP and emissions, and especially on the economic forecasts used. The national GHG emission target set by the EU (EU 2009b) is out of the studied range for Spain, Lithuania, Italy, Slovenia and Estonia, but close to the average range of the studied scenarios for most of the countries (Figure 4).

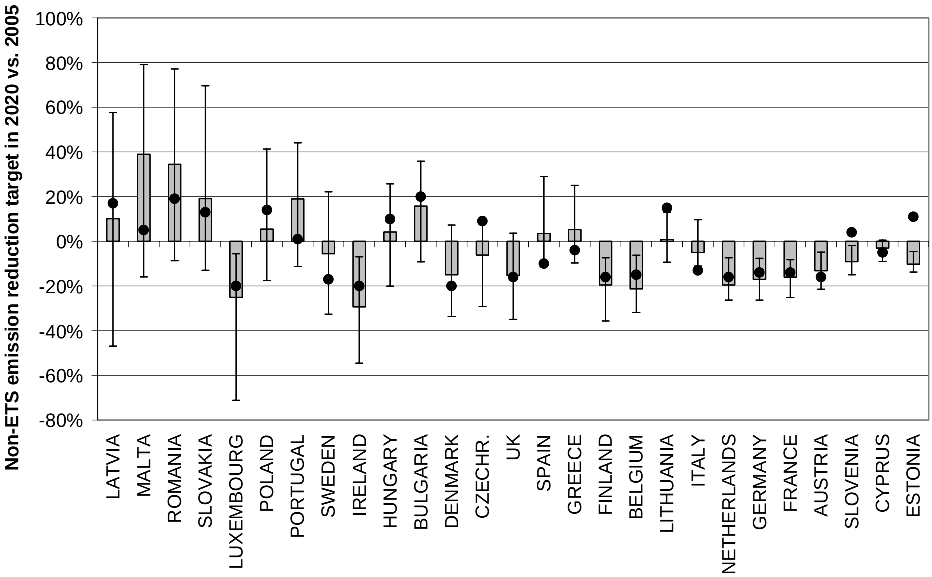


Figure 4. Average change in non-ETS GHG emissions by 2020 in comparison with 2005 using four different effort sharing criteria (adapted from Paper V). Error bars represent the variation range (min and max) in terms of percentage points of the criteria studied. The national GHG emission targets set by the EU (2009b) are illustrated by black circles. Countries furthest left have the largest variation between scenarios.

When looking at the requirements for improving the GHG intensity of economy in the non-ETS sector, the relatively fastest improvement was required in particular in Luxembourg, Ireland and some Eastern European countries, like Poland and Romania (Paper V). However, according to the scenarios, Ireland was the only country that came close to maintaining the historical rate on average. Latvia faced great GHG emission reduction requirements, if emissions were to be reduced based on reductions in GHG intensity in the past. Nevertheless, Latvia was allowed on average less improvement in annual GHG intensity than during 1993–2005. Slovakia, Romania and Poland faced the toughest GHG intensity reduction requirements in a scenario based on equal GHG per GDP criteria. For Sweden, UK, Finland and Denmark, the required effort was less than double the historical rate.

4.3.2 At the global level by 2050

Radiative forcing in 2100, calculated with the ETSAP-TIAM model, was 3.6 and 3.0 W/m^2 in target scenarios with the stabilization of the atmospheric GHG concentrations to 550 and 485 ppm CO_2-eq., respectively. The corresponding figures for the global mean temperature increase in 2100 were 2.1 and 1.8 °C. Depending on the emission reduction target scenario and the underlying socio-economic baseline scenario, the GHG emission allowances for Annex I[5] allocated by the Triptych and Multistage approaches varied from 10% to 50% reductions in 2020, and from 60% to 95% reductions in 2050 compared to the level of 2000 (Figure 5). Non-Annex I regions were allowed to increase their emissions up to 2020 by varying amounts, whereas in 2050 only the least developed regions received allocations above their 2000 emission levels. It should also be noted that the Multistage approach generally allocated more emissions to the least developed countries in 2050 than Triptych.

[5] Parties include the industrialized countries that were members of the OECD in 1992, plus the EIT countries, including the Russian Federation, the Baltic States, and several Central and Eastern European States.

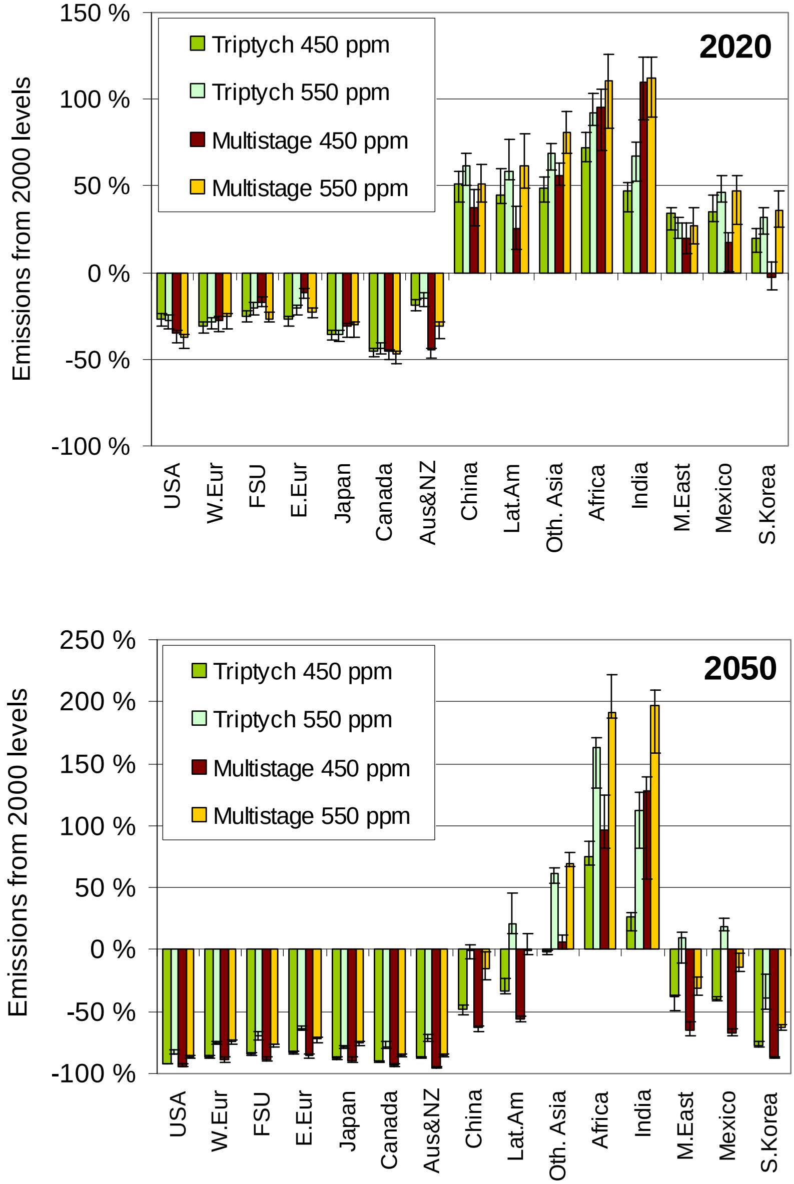


Figure 5. GHG emission allocation, relative to 2000 emissions, with the Triptych and Multistage effort sharing approaches with the 550 and 485 ppm CO_2-eq. stabilization targets in 2020 (up) and 2050 (down) (Paper VI). The error bars correspond to the range of values with four IPCC SRES baseline scenarios. (AUS&NZ = Australia and New Zealand, E.Eur = Eastern Europe, FSU = Former Soviet Union, Lat.Am = Latin America, M.East = Middle East, Oth. Asia = Other Asia, S.Korea = South Korea, W.Eur = Western Europe)

According to the analysis carried out using the ETSAP-TIAM model, the electricity sector provided the largest cost-efficient GHG emission reduction potential (Figure 6). The phase-out of coal and other fossil fuels with the large-scale adoption of wind power and bioenergy and also to some extent nuclear power and hydro power, and the use of combustible fuels in conjunction with CCS, contributed to most of the emission reductions. In addition, large emission reductions were made in the industrial sector and a number of measures were also introduced in the other sectors. The phase-out of fossil fuels and the use of CCS also played an important role in industrial emission reductions together with, among others, changes and improvements in industrial processes, such as an increased use of steel scrap or inert anodes in aluminium smelters and N_2O emission reductions using thermal destruction and catalytic reduction, respectively, in adipic and nitric acid industries. In road transportation emission reductions through a shift to natural gas, electricity/hydrogen and biofuels (when sustainably produced) were feasible. However, due to a rising demand for road and international transportation together with limited emission reduction potential for international transportation, the level of transportation emissions increased and remained approximately constant in the 550 ppm and 485 ppm scenarios, respectively. In agriculture the emission sources are very dispersed, often subject to major uncertainties and mostly concentrated on the rural areas of less developed countries. Consequently, it is difficult to control the emissions and effectively introduce enhanced practices, and thus only limited low-cost emission reduction potential is included in the model.

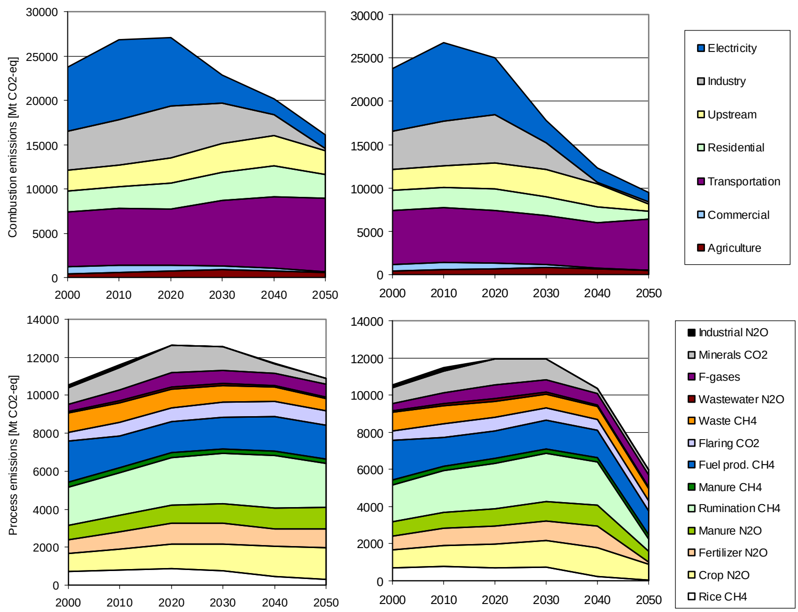
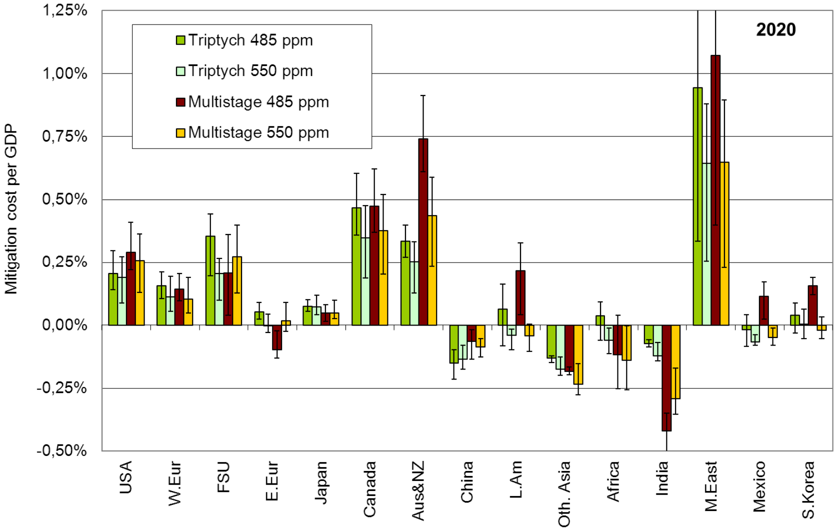


Figure 6. Global GHG emissions under the moderate growth B2 scenario with the 550 ppm (left) and 485 ppm (right) CO_2-eq. stabilization targets, split between combustion-based (top) and process-based (bottom) emissions (Paper VI). Non-CO_2 emissions converted to CO_2-eq. by using GWP-100 according to IPCC (1996).

The share of global emission reduction costs in GDP was approximately 0% in 2020 (less than 0.14% in all scenarios), and varied approximately from 1% to 2% and from 4% to 5% in 2050 in the 550 and 485 ppm CO_2-eq. scenarios, respectively, depending on the underlying socioeconomic baseline scenario. The marginal costs of emission allowances in 2050 rose as high as to 250–500 and 600–1000 USD2000/t CO_2-eq. in 550 and 485 ppm CO_2-eq. scenario, respectively.

Both Triptych and Multistage rules allocated costs for Annex I countries in 2020 (with the exclusion of Eastern Europe), costs around zero for more developed non-Annex I countries, and gains for least developed countries as a result of selling emission allowances (Figure 7). In 2050, Annex I countries, especially Australia and Russia (as a part of the former Soviet Union), faced relatively high costs in the 485 ppm CO_2-eq. target. Also, most non-Annex I countries faced positive costs, and only India and Africa were able to gain financially from the effort sharing. The costs for Annex I regions were generally doubled in the 485 ppm CO_2-eq. target in 2050 compared to the 550 ppm CO_2-eq. target. A clear outlier from the overall pattern with all effort sharing rules was the Middle East, in which the emission reduction costs arose to a large extent from lower revenues from oil trade.



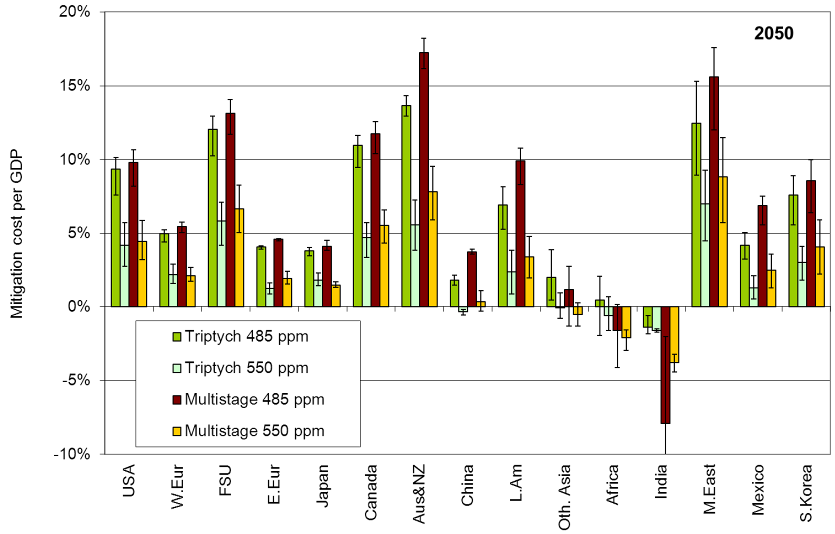


Figure 7. Regional GHG emission reduction costs relative to their baseline GDP in 2020 (up) and 2050 (down) (Paper VI). The error bars correspond to the range of values with four IPCC SRES baseline scenarios. (AUS&NZ = Australia and New Zealand, E.Eur = Eastern Europe, FSU = Former Soviet Union, Lat.Am = Latin America, M.East = Middle East, Oth. Asia = Other Asia, S.Korea = South Korea, W.Eur = Western Europe.)

5. Discussion

5.1 Attributing emissions and emission allowances

Attributional life cycle assessment (ALCA) aims to describe a product system as it is without aiming to capture the consequences of introduction, modification or decommissioning of the product system. Similarly, various criteria to differentiate emission reduction commitments at country or country group level such as Triptych and Multistage approaches aim to attribute emission allowances between the countries or country groups without regard to the consequences *per se*. The quality of data and criteria to attribute the potential emissions or environmental impacts and emission allowances in ALCA and effort sharing, respectively, are the critical underlying issues influencing the outcomes.

5.1.1 Emissions at product level

The sustainability criteria of the EU Renewable Energy Directive (RED) require, among others, determination of GHG emission reduction of transportation biofuels and other bioliquids compared to reference fuels. This should take place prior to or after the production of a certain quantity of the products. This can be done under certain conditions by using the default values given in the RED or by calculating the actual GHG emission saving compared to the reference fuels by using the given methodology. The assumptions used in determining the default values in the RED (BIOGRACE 2012) and in the specified RED methodology to calculate actual GHG emissions mostly follows the principles of ALCA as analysed in Paper II.

As presented in Papers I and II, the uncertainties of GHG emissions of biofuels may be very significant. Regional differences clearly create natural variation in results between different studies. For example, the GHG emission intensity of RME studied in Paper I was higher than in most of the studies reviewed by Malca and Freire (2011). Only a few studies (Reijnders and Huijbregts 2008, Harding et al. 2007) have arrived at a GHG emission intensity of the same magnitude as that presented in Paper I. The yield per nitrogen fertilizer requirement is relatively low in Finland mainly due to climatic conditions influencing, for example, the growing season, nitrogen transfer from soil to plants and thus also feasible plants (Peltonen-Sainio and Jauhiainen 2010, Peltonen-Sainio et al. 2007). For example, the typical ratio of yield per N-fertilizer use in Finland for oil plant (spring turnip rape in Paper I) cultivation is approximately 16 kg/kg, whereas it equals roughly 22 kg/kg in the EU-25 on the average (JEC 2011) and roughly 20 kg/kg in Southern Sweden (Ahlgren et al. 2009). However, regional differences only explain part of the differences in GHG emission figures. Significant variation may also take place, for example, due to the way the parameters are determined and considered. In a review of a number of studies concerning GHG emissions of RME in Europe, Malca and Freire (2011) noted that treatment of co-products and land-use modelling including N_2O and CO_2 emissions from soils were the key issues resulting in significant variation between the studies. Similarly, in Paper I, N_2O emissions from soil, GHG emissions from electricity production and soil carbon changes due to raw material harvesting were recognised as being particularly important. Comprehensive screening of the differences between various studies is challenging and would require detailed meta-analysis.

Deterministic default values of the RED do not include any uncertainty range, as presented in Figure 2. In addition, in May 2012 it was unclear how required parameters and the involved uncertainty are to be considered in the accounting of actual GHG emissions in the context of the RED. The default values of the RED exclude carbon stock changes in soil and terrestrial biomass (BIOGRACE 2012) and they are not specifically obliged to be included in the calculations of actual GHG emissions when land-use change from one land use class to another does not take place (EU 2009c). The exact determination of parameters is not specified in the RED, except for the general frames for emissions to be accounted and the fixed rule for allocation (EU 2009c) as well as information for accounting for land carbon stocks in the case of direct land-use changes (EC 2010c). This may lead to significant differences in the determination of the actual GHG emission saving values of various biofuel chains.

Emissions are always generated in comparison to some reference situation. Typically, in ALCA the reference level is the absence of the use of resources (“no use”) generating the emissions (e.g. fossil fuels). However, regarding land use the reference situation is dynamic. According to a framework for LCIA of land use released within *The UNEP-SETAC Life cycle initiative* (Milà i Canals et al. 2007), in ALCA the “no use” reference situation is the natural relaxation of the land area. In practice, the determination of GHG emissions from the “no use” reference situation should always be based on assumptions which cannot be measured or monitored, creating an element of uncertainty. The determination of the reference situation for land use is not specified in the RED.

GHG emission reductions are often measured in relative terms compared to a reference functional unit (e.g. the use of fossil fuels to produce the same functional unit). In many recent studies concerning biofuels, and in the RED methodology, the relative emission reduction indicator is determined as the difference of the GHG emission balance between the fossil reference fuel and the biofuel compared to the fossil reference fuel (see Equation 1). The fundamental problem of this particular kind of ‘relative emission reduction’ indicator is the inability to measure the effectiveness of biomass utilisation as a measure to reduce GHG emissions.

The relative GHG emission savings may look particularly favourable for biofuel processes in which significant amounts of low GHG emission intensive raw materials are used in relation to the amount of biofuel produced. At the same time, another process for converting biomass to biofuel in a more energy-efficient way, while using more fossil resources, may appear unfavourable in terms of the particular indicator. The effectiveness of use of the limited resources – biomass and land – is excluded when using this kind of 'relative emission reduction' indicator. Consequently, this particular indicator cannot be used to compare GHG emission reductions between different end-use options for biomass, for example transportation biofuel and electricity production. In order to promote the most efficient options of biomass and land use in climate change mitigation, other kinds of 'relative emission reduction' indicators may be more appropriate. It would be reasonable to measure the GHG emission balances or savings of biofuels in terms of the limiting factors, for example biomass, land area or money spent (Schlamadinger et al. 2005). 'The relative emission reduction' indicator presented in Figure 1 takes into account the biocarbon consumed for the emission reduction.

The determination of GHG emissions is a key issue concerning electricity consumption of product systems in ALCA, for example in the production of biofuels in the context of the RED. As presented in Section 4.2, the annual variation, selection of allocation method and consideration of electricity trade between countries significantly influence the annual average CO_2 emission intensity of electricity in many countries. In Papers III and IV, the use of the consumption-based method is advocated in preference to the production-based method for LCA purposes. However, the use of one allocation method as superior to others cannot be suggested based on the results of Paper IV. As presented in Section 3.1.1, the allocation should primarily be avoided whenever possible or be based on physical causal relationship of the products. If this cannot be done, the allocation can be based on other relationship of the products. As physical causalities cannot be determined to CHP plants, which are built to jointly produce electricity and heat (Frischknecht 2000), a non-causal-physical relationship needs be used as a basis for allocation. In Paper IV, allocation based on energy content and 'motivation electricity' was selected to represent the lower and the upper boundary of the range, respectively, of the CO_2 emission intensity of electricity. Both of these methods are applied in practice. In the RED methodology allocation is determined to be based on lower heating value of the products in case no biofuel production is related to the electricity production. Regarding CHP, this probably means the use of 'the motivation electricity' method as heat does not have lower heating value. On the other hand, allocation based on energy content of the products is suggested for CHP in the *Energy Statistics Manual* jointly produced by IEA and EUROSTAT (IEA 2004). As presented in Papers III and IV, the use of only one allocation method may be highly misleading. When allocation cannot be avoided, and if only one particular allocation method is to be applied, an allocation based on economic value is suggested as the most suitable option (Guinée et al. 2004). In addition, Ekvall et al. (2005) concluded that allocation should be based on the economic value of the products when the aim of the study is to describe the causes of the environmental burdens of the life cycle in ALCA. The allocation method presented in the RED is not consistent with these conclusions.

The figures presented in Section 4.2 for CO_2 emission intensities of electricity consumption do not include upstream emissions from supply of the fuels and production of the infrastructure and power plants. These, however, typically constitute a relatively low share of GHG emissions of the overall electricity production mix (e.g. Kim & Dale 2005, Santoyo-Castelazo et al. 2011, Lee et al. 2004), although for certain power production technologies they may be significant (Frischknecht et al. 2007, Weisser 2007). However, an extensive shift in energy production systems may occur within the next few decades with the large-scale introduction of low GHG emission intensive power production technologies as a result of ambitious climate change mitigation targets (IPCC 2007c). Consequently, in the overall life cycle of electricity consumption, the contribution of GHG emissions other than direct CO_2 emissions from fuel combustion might increase significantly, and would therefore need to be considered more carefully. In particular, GHG emissions related to the cultivation and harvesting of bioenergy have already been widely discussed. Also, CH_4 and N_2O emissions from fuel combustion should be considered and they may play relatively significant role for some combustion technologies (Tsupari et al. 2005, 2007).

The definitions of spatial and temporal system boundary for the electricity production mix are crucial issues. Apart from annual national average mixes, smaller or larger regions and shorter and longer time frames may also be selected. As discussed in Paper III, figures based on the contract between the electricity seller and the customer with real-time accounting would be the ideal production mix figures for history-related ALCA. A general introduction of this kind of 'contract-based' approach would eliminate the prevailing problem in selecting the spatial and temporal dimension arbitrarily. Currently, such data and respective reporting practices do not generally exist, and thus further research and agreements between various stakeholders are required. For future-related ALCA studies, the development of the power production system should be considered by using an appropriate scenario analysis.

Ideally all environmentally relevant physical flows from the cradle to grave of a product system are included in ALCA. In practice it is constrained by time and resource limitations, and parts of the system, such as services and capital goods, are usually ignored or cut off from the analysis. The impacts of the neglected parts on the GHG emission results may vary significantly depending on the system (Suh et al. 2004, Ferrao & Nhambiu 2009, Mongelli et al. 2005, Mattila et al. 2010). Approaches to consider potential environmental impacts of flows which are not necessary included in LCA based solely on process description (process-LCA) are so called input-output-LCA (IO-LCA) without using any process-based life cycle inventories and hybrid-LCA combining both process-LCA and IO-modelling (Suh 2004, Suh & Huppes 2005, Hendrickson et al. 2006). The question whether available databases of IO with environmental extensions are robust enough has been raised and progress to improve the quality and applicability of the data is being made in various countries (Finnveden et al. 2009).

5.1.2 Emission allowances at country level

The effort sharing of national (non-ETS) emission targets of the EU Member States in 2020 were studied in Paper V. Unanimous annual reduction, historical development and convergence in GHG/GDP as well as GHG/capita convergence were applied as a basis for sharing emission targets. The emission reduction requirements for a given country varied significantly depending on the criterion applied, which confirms the findings of den Elzen et al. (2007). Furthermore, changes in underlying assumptions, such as the selection of the base year applied, the allocation of GHG emission reductions between the ETS and non-ETS and the choice of GDP forecasts, as studied in the sensitivity analysis in Paper V, posed significant variation in the results.

Triptych and Multistage approaches were studied for global effort sharing in Paper VI. Both approaches allocated emission reductions to the 15 regions studied very differently, in particular for non-Annex I countries. In general, compared to Triptych, the Multistage approach allocated clearly more emission allowances to the least developed countries due to assumed later participation in the binding commitments. The baseline scenario and the overall emission reduction target also significantly influenced the results. Also, the accuracy related to historical GHG emissions applied as a basis for assumed future baseline emissions of Triptych and Multistage played an important role. Using different historical emission estimates (e.g. change from the UNFCCC data to IEA/EDGAR data) might imply differences of several tens of percentage points on the allowances a country receives (Paper VI). Furthermore, the other assumptions used in Triptych and Multistage approaches to set emission reduction targets for the countries certainly influences the results, although this is not studied in Paper VI. For example, Soimakallio et al. 2006 concluded that, although sensitivity analysis carried out for the Triptych 6 and Multistage approaches for some methodological assumptions indicated a relatively low variation compared to the impact of baseline scenario, more methodological changes might have resulted in more significant variation. The recalibration of the EVOC tool that was carried out in Paper VI resulted in large changes in the emission allowances allocated by the Triptych to certain countries, especially for Australia in 2050, highlighting clearly the importance of assumptions used in the effort sharing process.

5.2 Capturing consequences

Consequential life cycle assessment (CLCA) aims to describe at product system level how environmentally relevant physical flows would have been, or would be, changed in response to possible decisions that would have been, or would be, made. Similarly, bottom-up modelling can be used to assess consequences taking place at sector, national or global level due to various decisions, such as targets to mitigate climate change and emission reduction effort sharing. For both types of assessment of consequences a number of assumptions are required. The fundamental problem is the difficulty in identifying the change from the reference scenario due to a complex cause and effect relationships.

5.2.1 Increased production of biofuels

The analysis of Paper I followed the principles of CLCA. (However, Malca and Freire (2011) classified the method used in the particular paper as ALCA with no specified explanations). The results for GHG emission reduction of replacing reference fuels by biomass-based transportation fuels, electricity and/or heat in Finland reflected significant parameter uncertainties. Nitrous oxide emissions from soil, soil carbon losses, emissions from electricity production and emission reduction from replaced electricity were the most significant parameters, depending on the biofuel chain considered (Table 2). The uncertainties in other individual parameters had a clearly minor influence on the overall uncertainty range. The type of probability distributions were selected subjectively in Paper I, and the uncertainty due to that selection was not studied. Instead, Plevin et al. (2010) tested a range of various types of probability distributions, and concluded that the shapes of the probability distributions studied had relatively little effect on the shape of the output frequency distribution in their case study. However, this conclusion cannot be directly applied to the analysis carried out in Paper I, and should therefore be studied.

Also, the other assumptions used in CLCA are of central importance. In Paper I, it was assumed that land and raw materials were available for biofuels. However, this is not necessarily the case in practice. As discussed in Paper II, the taking of agricultural land for biofuel raw material production may transfer other agricultural activities indirectly elsewhere. The consequences may be very far reaching in space and time, including deforestation and significant carbon dioxide emissions (e.g. Searchinger et al. 2008, Plevin et al. 2010, Edwards et al. 2010). There is support for the assumption that an increase in soy in, for instance, Mato Grosso, Amazonia, has displaced pasture, leading to deforestation elsewhere (Barona et al. 2010). According to IPCC (2011), the significance of land-use changes (LUC) on GHG emissions of products was demonstrated in the 1990s when direct land-use changes (dLUC) effects were introduced in some life cycle assessment (LCA) studies (e.g. Reinhardt 1991, DeLucchi 1993). However, most LCA studies have not considered indirect land-use changes (iLUC) taking place through market mechanisms (IPCC 2011).

In recent years, a number of studies aiming to analyse dLUC and iLUC related to the increasing production of biofuels have been conducted. The simplest approaches to estimating predicted iLUC are based on aggregated recent historic data on biofuel feedstock determination and agricultural expansion, combined with assumptions on a number of crucial future-related parameters such as feedstock, co-product availability, likely LUC types and the associated lost carbon stocks (Cornelissen et al. 2009). Such approaches include the ones presented by Fritsche (2007), Ecometrica (2009), Scott-Wilson (2009) and Overmars et al. (2011). Over the past few years, the quantification of iLUC related to biofuels has mainly been carried out using various types of economic and environmental models jointly (e.g. Searchinger et al. 2008, Al-Riffai et al. 2010, Birur et al. 2008, Fabiosa et al. 2010, Edwards et al. 2010, Plevin et al. 2010). General scientific consensus exists on using an economic approach to address iLUC, but the methods are generally controversial (Kim & Dale 2011, O'Hare et al. 2011, Kline et al. 2011, Gnansounou et al. 2008). The results of an economic approach are highly sensitive to the assumptions used. For example, Barona et al. (2010) concluded that the drivers of Amazon deforestation need further research on how interlinkages between land area, prices and policies influence cultivation and deforestation. Furthermore, improvement of land-use modelling in PE energy system models and GE economic models, or more integrated modelling using such models and land-use models together are required to better assess the consequences related to expanding biofuel production. Plevin et al. (2010) concluded that, although the emissions from iLUC are subject to significant uncertainties, the emissions take place and there is a significant likelihood of large emissions.

Additionally, the competition of forest-based raw materials may cause remarkable indirect impacts. Forsström et al. (2012) concluded, based on partial equilibrium energy system modelling, that the introduction of large-scale production of transportation biofuels from forest-based raw materials in Finland would lead to significant re-allocation of wood use from other energy production and industry, thus increasing the use of other fuels in those sectors. Furthermore, they concluded that re-allocation of wood use from electricity and/or heat production to transportation biofuel production would result in an increase in GHG emissions in Finland. This emphasises the conclusion drawn, for example, by Ohlrogge et al. (2009) that greater reductions in GHG emissions can be achieved by using raw materials for power or heat production to substitute coal than by producing more energy intensive liquid biofuels to substitute oil.

Apart from the spatial dimension, also the temporal dimension of a system boundary is critical. In static temporal assessment, all GHG emissions and sinks are assumed to take place at the same time and they are then equalised over the lifecycle studied, resulting in model uncertainty. The exclusion of dynamics of the GHG emissions, sinks and avoided GHG emissions is problematic, particularly when they differ significantly over time, which may be the case for many bioenergy options (Kendall et al. 2009, Cherubuni et al. 2011). This is the case in particular when significant pulse emission takes place due to immediate land-use change (Kendall et al. 2009), or relatively slowly grown forest biomass is used (Pingoud et al. 2011). In Paper I the soil carbon losses due to logging residue harvesting were considered by estimating the amount of carbon that would have been accumulated into soil after 100 years in a reference situation. Even though capturing one dynamic dimension in Paper I, the particular approach does not take into account the fact that the carbon dioxide released from biofuel combustion compared to the reference situation is to be accumulated in the atmosphere, resulting in positive radiative forcing. Capturing the particular effect by using dynamic indicators such as those presented by Kirkinen et al. (2008, 2010) or derivates of them (e.g. the one presented by Pingoud et al. 2010, Repo et al. 2011 or Kujanpää et al. 2010), would result in an increase in the GHG impact of soil carbon losses over 100 years by approximately 30% compared to the figure applied in Paper I (Kujanpää et al. 2010). Furthermore, different time frames result in different conclusions. For example, applying 20 or 50 year timeframe results in significantly greater impacts compared to applying 100 year timeframe (Kirkinen et al. 2010, Pingoud et al. 2011, Repo et al. 2011, Kujanpää et al. 2010). The fundamental problem is that there exists no unique scientifically defined robust timeframe, rather the temporal dimension is a value-based issue reflected by the emphasis of contemporary climate policy.

In Paper II, the suitability of the RED methodology for ensuring GHG emission reductions of increasing production and the use of transportation biofuels and other bioliquids in practice are analysed and discussed. In the RED (methodology), all types of indirect effects through market mechanisms and the possible losses in soil and temporal carbon stocks are excluded in the determination of the default values and in the methodology to calculate actual GHG emissions. Consequently, there is a serious risk that the sustainability criteria of the RED underestimate the GHG emission impacts related to large-scale biofuel production and may promote biofuels with low reduction or even an increase in the overall GHG emissions and prevents biofuels with higher benefits at the same time.

5.2.2 Grid electricity consumption

Regarding electricity consumption or conservation in CLCA, the major challenge is to identify the marginal technology, and furthermore, the consequences influenced by the change (Paper III). In its simplistic form, marginal production, affected by the marginal change in the electricity consumption, is identified. Large variations between the affected technologies may occur. Using fundamentally different kinds of affected technologies for this kind of analysis has been suggested (Mathiensen et al. 2009). However, the instant marginal GHG emissions of electricity production do not reflect the market effects beyond the immediate change. Such effects may take place in the short term (e.g. increases in electricity price) and long term (e.g. investment decisions). The anticipated development of energy prices, quantity and time-dependent profile of electricity consumption as well as climate policy are probably the most important market drivers of new investments in electricity production (Lund et al. 2010). The range applied for GHG emissions of marginal electricity consumption (0–900 g CO_2-eq./kWh) in Paper I fits quite well with the long term marginal technology mix presented by various papers cited and discussed in Paper III. Furthermore, the range (300–900 g CO_2-eq./kWh) applied to electricity consumption replaced by biofuels in Paper I can be justified by the fact that the targets for increasing the use of renewable energy sources in the EU are so massive that it is very unlikely that the use of renewable energy sources will be replaced by bioenergy. Thus, the lower limit can be considered to reflect the replacement of the use of the low GHG emission intensive fossil fuel that is relatively efficient natural-gas-fired condensing power.

As changes in the power system are not isolated, electricity consumption and production cannot be separated from one another (Lund et al. 2010). When attempting to study the consequences of a decision to change electricity consumption on GHG emissions, an improved understanding of the phenomenon is certainly required. It is important to recognize that, not only the electricity production system is affected, but probably many other economic activities as well. Scenarios that depict the changes in economic inputs and outputs can be constructed using economic equilibrium models (e.g. Manne et al. 1995, Nordhaus 1999, Nijkamp et al. 2005). Yet, due to the complexity of such models, the energy system is typically described in relatively rough terms, limiting the suitability of such models for assessing, for example, GHG emission impacts. Partial equilibrium models for energy systems such as ETSAP-TIAM used in the analysis of Paper VI and others presented e.g. in Lund et al. (2010) and Klaassen & Riahi (2007) can provide detailed information on the development of energy production in supplying external energy demand. By using economic equilibrium and partial equilibrium models simultaneously, it is possible to create far-flung scenarios to determine the development of GHG impacts of the economies and various actions. Yet, scenarios always involve a certain degree of uncertainty. Consequently, it is suggested that an appropriate number of scenarios are carried out for CLCA in order to provide adequate perspectives on the evolution of the economies, electricity consumption and production as well as GHG emissions under various relevant market conditions.

5.2.3 Costs of effort sharing

The direct impact of emission reduction effort sharing is the distribution of the emission reduction costs between the countries. In Paper V costs resulting from the application of various effort sharing scenarios studied were not considered. In Paper VI the economic burden of emission reductions was shared through the allocation and trade of emission allowances. Thus, the price of allowances became a critical factor for the costs the countries faced. Besides depending on the effort sharing the price of allowances also depends on the direct emission reduction costs. The baseline scenario and descriptions of cost-curves and potentials of technologies furthermore affected the marginal abatement curve (MAC) of a country. This can be noted by reflecting the results presented in Paper VI to other comparable studies (e.g. den Elzen et al. 2008b, van Vuuren et al. 2007). The global costs between the studies were quite similar, but the marginal costs in comparable studies were lower compared to those presented in Paper VI, in particular due to more pessimistic assumptions used for non-CO_2 emission reduction and bioenergy supply potentials in the ETSAP-TIAM model. Uncertainties of MACs are much larger in the more ambitious 485 ppm CO_2-eq. scenario, in which more unconventional emission reduction measures have to be taken in order to reach the emission target compared to 550 ppm CO_2-eq. scenario. The effect of technological and resource uncertainties on effort sharing might, however, be minor, as most technologies affect all countries (den Elzen et al. 2005). On the other hand, den

Elzen et al. (2008b) noted that a specific technology cost, CCS's in their case, might affect some countries more than others. Allowance prices might also carry additional uncertainty due to market imperfections as studied in the sensitivity analysis of Paper VI.

The partial equilibrium approach used in Paper VI, while providing a detailed picture of the direct emission reduction costs, does not include any feedback effects from the rest of the economy. Effort sharing, especially in the extreme cases, might involve large wealth redistributions through allowance markets, affecting affluence levels and energy demand. Furthermore, a high price of emissions is likely to induce structural change in the economy. Should the demand and production structures adjust to the cost of carbon, the mitigation costs would then be lower than reported here. With the ETSAP-TIAM model, the only possible adjustment is reduced demand (i.e. welfare loss) instead of, for example, demand substitution. What is more, the avoided damage costs from climate change through mitigation were ignored. To provide a broader picture of the costs and avoided costs, wider economic and risk assessment analyses are required through CBA.

5.3 Avoiding emission leakage

GHG emission leakage takes place when the consumption of goods and related production are geographically separated. Weak definition of leakage considers the total aggregated GHG emission flows embodied in trade, typically from non-Annex B to Annex B countries with binding emission reduction targets under the Kyoto Protocol (Peters & Hertwich 2008). Strong carbon leakage is used when policy change in an Annex B country causes production to increase in a non-Annex B country (ibid.). According to Peters et al. (2011), the net CO_2 emission transfers from developing to developed countries exceeded the GHG emission reduction targets of the developed (Annex I) countries in the Kyoto Protocol.

Global commitment into country-specific emission caps as studied in Paper VI would significantly reduce or even avoid the risk of emission leakage. Even though developing countries were allowed to increase their emissions in 2020 and the least developed countries even in 2050, the commitment to a cap-and-trade system would prevent the possibility of unlimited emission growth in non-Annex B countries. However, as there is no agreed systematic approach for effort sharing, for example based on certain criteria, under the UNFCCC, the international climate negotiations are completely dependent on pledges given by the countries. The risk of significant GHG emission leakage between countries exists at least as long as a comprehensive and effective climate convention is lacking.

One solution for reducing significant emission leakage could be the introduction of consumption-based emission targets for countries or products based on an end-use responsibility point of view (Pingoud et al. 2010). The sustainability criteria for transportation biofuels and other bioliquids of the EU are an example of this kind of approach. However, exclusion of indirect impacts from the system boundary considered, as in the case of the EU RED, would not remove the problem of emission leakage. One option for reducing indirect impacts could be the use of certain types of wider average data instead of case-specific data, for example related to land-use changes, as suggested in Paper II and by Saikku et al. (2012). In such an approach, indirect impacts are moved from the consequential framework to be an attributional issue by extending the system boundary for emission attribution. Another option would be the extensive introduction of consumption-based criteria not only for certain applications such as biofuels but for various products. Ideally, if all the products were monitored, no unmonitored indirect impacts would take place. However, consumption-based determination of emissions encounters the problems of life cycle assessment, which makes it difficult to find a consensus between a number of parties or stakeholders as to the practical solution. In addition, the countries that are not ready to take binding national emission caps would be unlikely to commit their industry to binding consumption-based targets either.

5.4 Equity issues

Different types of perspectives on equity are encountered in LCA and emission reduction effort sharing. Fundamentally, there is a dilemma between undesirable environmental consequences and responsibility. In LCA, there is a need to select between an attributional and a consequential approach and the related system boundaries, between average and marginal data, and between various allocation methods. In effort sharing, the criteria and data to be applied need to be defined. The selections may be considered fair or unfair from various points of views.

The technical limitations of subjective choice of system boundary setting and other methodological choices in LCA have equitability implications. For example, the cut off rule to exclude the emissions from the construction of machinery and infrastructure and the rule not to allocate emissions to co-produced heat, applied likely in the EU RED methodology, may be considered unfair to fuel producers or other stakeholders, especially if they would have played an important role in the GHG emission reduction results of a product. Arbitrary determination of appropriate average data to be used in ALCA is also problematic. The use of average data instead of case specific data, for example related to the determination of appropriate electricity production or land-use mix, may be unfair to those actors doing significantly better environmentally than the average level. On the other hand, the use of case specific data may be considered unfair to those actors not having an opportunity to use the particular resource, as it includes an assumption of the right to use certain resources regardless of their availability. CLCA is subject to inherent uncertainty, as it is not possible to consider all the impacts and the uncertainty in the marginal effects increases with the time horizon.

Apart from technical limitations, both ALCA and CLCA also have endogenous ethical limitations. According to Ekvall et al. (2005), ALCA (retrospective in their typology) is consistent with both deontological[6] and teleological[7] rule ethics, whereas CLCA (prospective in their typology) is valid from the perspective of teleological situation ethics. The RED sustainability criteria for transportation of biofuels and other bioliquids seem to reflect a special case of deontological and teleological rule ethics. The rule adopted in the criteria does not have links to all of the consequences (e.g. indirect impacts), but is introduced so as not to be associated with systems that have undesirable climate impacts (e.g. direct deforestation). If the RED sustainability criteria were modified to better include the consequences, for example iLUC (EC 2010d), this could be an example of how CLCA generates the information that is relevant in the context of teleological rule ethics. ALCA includes a risk of unaccounted undesirable consequences, whereas CLCA holds a risk of unfair results and suboptimised systems (Ekvall et al. 2005), raising the question of the responsibility of the marginal effects. One example is the question of whether the 'new electricity consumption' should be considered differently (e.g. by using marginal data) from 'the existing one', and if so, what are the implications of using this information in decision-making.

The choice between an attributional and a consequential approach is significant, though from a certain point of view they can both be considered equitable and legitimate. When aiming to avoid life cycles and subsystems that have an undesirable environmental impact, ALCA is useful in decision making. Similarly, if the changes in product systems are considered 'good' if consequences for the total environment are lowered, then CLCA is valid (Ekvall et al. 2005). From the perspective of utilisation of LCA results by, for instance, consumers or policy-makers, it can be considered unfair if the results are not reported in the light of goal and scope of the study. The major uncertainties and sensitivities involved, as well as the limitations of the applicability of the results, should be reported. The goal by definition in LCA should not be to assess everything exactly at the most detailed level, but to create relevant information for decision-making.

Equity is a fundamental but also an ambiguous issue in emission reduction effort sharing. For example, Ringius et al. (1998) define five different equity concepts: 1) Egalitarian (equal emissions per capita), 2) Sovereign (equal emission reductions from e.g. 2000), 3) Horizontal (equal net change in welfare e.g. in GDP), 4) Vertical (effort depending on ability), 5) Equal responsibility (effort based on historical emissions). Different effort sharing criteria follow different equity principles and result in different implications. Ultimately, the effort sharing under the UNFCCC will be a result of political climate negotiations in which a systematic effort sharing approach may either be used or not. There is no definitive answer to the equitable balance between the costs and gains of different parties, but a quantified assessment of possible outcomes might aid the process considerably. One

---

[6] The normative ethical position that judges the morality of an action based on the action's adherence to a rule or rules.

[7] Ethical theory that holds that the consequences of an act determine whether an act is good or bad.

major problem is that the costs can be assessed by using various assumptions concerning, for example discounting and exchange rates of currencies, and from very different perspectives, including or excluding social costs, which are very important but typically subject to significant uncertainties compared to direct costs (Tol 2003). On the other hand, if a consensus in effort sharing is found, it could be considered to be an equitable solution.

5.5 Climate impacts, sustainability and multi-criteria decision-making

In this study, GHG emissions, avoided GHG emissions and associated direct costs were considered as well as climate impacts in terms of global mean surface temperature increase. Other possible types of climate impacts such as sea level rise, floods, droughts and diseases were excluded, as well as other types of impacts influencing radiative forcing such as albedo changes through land-use changes, aerosols and black carbon on snow. These issues may be very important but they are also subject to remarkable uncertainties (IPCC 2007a, b). Furthermore, climate sensitivity to increasing concentrations of GHGs is highly uncertain (IPCC 2007a). Consequently, more information is required in order to more reliably assess overall warming and follow the climate impacts of various measures or emission paths.

As discussed in Section 5.2.1, in the context of carbon stock changes, the time frame in which the climate impacts or climate change mitigation are considered is highly relevant. Typically, various non-CO_2 GHG emissions are characterized as carbon dioxide equivalents by using GWP-100 factors, which are officially used in annual GHG emission reporting to the UNFCCC and the Kyoto Protocol. However, the time frame is critical when weighting cumulative radiative forcing of different GHGs, as they have significant differences in their specific infrared absorption properties and atmospheric lifetimes which are, furthermore, subject to uncertainties (IPCC 2007a). For example, the use of 20-year time horizon instead of 100 years roughly triples the global warming potential of CH_4, whose atmospheric life time is only some 12 years. Furthermore, the uncertainties of the direct GWP factors provided by the IPCC are estimated to be ±35% for the 5 to 95% (90%) confidence range. (Ibid.)

Apart from GWPs with various time frames other types of metrics have also been proposed to characterize various GHG compounds. The global temperature change potential (GTP) is a physical metric that compares the global average temperature change at a given point in time resulting from equal mass emissions of two greenhouse gases (IPCC 2009). As the assumptions on climate sensitivity to radiative forcing and the exchange of heat between the atmosphere and the ocean are included in GTP, greater uncertainty is involved in the particular metrics compared to GWP. Substantial work has also been performed on metrics that combine physical and economic considerations, such as global damage potential (GDP) and global cost potential (GCP) (IPCC 2009).

When comparing the emissions of gases with substantially different lifetimes, the choice of metric becomes very important. Compared to CO_2 emissions, the choice of metric has much greater implications for CH_4 than for N_2O, whose atmospheric lifetime is more akin to the lifetime of CO_2 (IPCC 2009). No single metric can accurately consider and compare all the consequences of the emissions of different GHGs. Thus, the most appropriate metric and time frame depend on the purpose and aims of climate change mitigation, which may, for example, be the limitation of global equilibrium surface temperature increase, limitation of global surface temperature gradient or limitation of instant surface temperature.

Apart from climate impacts, sustainability is a broader issue which has environmental, economic and social dimensions. Sustainability is a capacity to endure, which means for humans the long-term maintenance of responsibility. According to the most quoted definition, sustainable development (currently usually known as sustainability) “is development that meets the needs of the present without compromising the ability of future generations to meet their own needs” (WCED 1987).

As regards the environmental dimension, sustainability requires methods and tools to measure and compare the environmental impacts of human activities for the provision of goods and services (Rebitzer et al. 2004). Human actions constitute a diverse range of emissions and resource consumption contributing to a wide range of impacts, such as climate change, stratospheric ozone depletion, tropospheric ozone (smog) creation, eutrophication, acidification, toxicological stress on human health and ecosystems, the depletion of resources, water use, land use and noise (Rebitzer et al. 2004). Today there is acceptance in the LCA community that the protection areas of Life Cycle Assessment are human health, natural environment, natural resources and to some extent the man-made environment (Udo de Haes et al. 1999, 2002). Impacts on the areas of protection are modelled applying the best available knowledge about relationships between interventions in the form of resource extractions, emissions, land and water use, and their impacts in the environment (Finnveden et al. 2009). A distinction is made between midpoint and endpoint, where endpoint indicators are defined at the level of the areas of protection, and midpoint indicators indicate impacts somewhere between the emission and the endpoint. Endpoint modelling is more reliable for certain impact categories such as acidification, cancer effects and photochemical ozone formation, while it is still under development, for example for climate change due to large uncertainties and the long time horizons of the endpoint (Finnveden et al. 2009). In addition, certain impact categories may include several types of impacts. An example is land use which can be separated among others into loss of biodiversity, loss of soil quality and loss of biotic production potential (Milà i Canals et al. 2007, Udo de Haes 2006).

Utilisation of LCA results in decision making requires the weighting of various environmental indicators. Furthermore, in many real life situations, LCA results are not the only criterion on which the decision is made. As regards sustainability as a whole, economic and social dimensions should also be taken into account and be weighted towards each other and various environmental indicators. Work has been done to integrate the three dimensions of sustainability through development and analysis of various methods such as life cycle costing (LCC) and social life cycle assessment (SLCA) (see e.g. CALCAS 2009). Weighting requires the inclusion of social, political and ethical values which are influenced by the perception of outcomes from science.

Multiple criteria decision analysis (MCDA) is used in the weighting of various indicator results into an overall sustainability score (Finnveden et al. 2009). In MCDA, the utility model consists of multiple decision criteria with subjective weights describing the relative importance of the criteria and decision alternatives and their performance with respect to each decision criterion (e.g. Saaty 1980, Keeney & Raiffa 1993). The decision-making problem depends on the uncertainty of LCA indicators, but also significantly on the weighting of the indicators and the related uncertainty (Mattila et al. 2012). In general, it cannot be determined whether the uncertainty of a single LCA indicator is significant, and whether the LCA is adequately reliable or not. For example, the choice from among various production methods for a product depends on the uncertainty level, the difference in the average utility ratios of the alternatives and the attitude of the decision-maker to risk (ibid.). It is possible that the weighting issues should be decided upon in advance, since it is not necessarily meaningful to carry out detailed, complex, comprehensive and probably costly uncertainty analysis if the relevant LCA indicator is given low weight in decision-making (ibid.).

6. Conclusions and recommendations

This study showed that there are significant uncertainties involved in the GHG emissions of biofuels and grid electricity consumption at product level and in the effort sharing of GHG emission reduction commitments at country or country group level. Parameter variation and stochastic simulation, successfully used in this study, are valid methods for propagating parameter uncertainties. However, the results provided by such methods should not be overinterpreted, as the results of any life cycle assessment (LCA) or effort sharing are only valid with the assumptions made.

Scenario analysis and parameter variation related to methodological choices needs to be carried out in order to understand the importance of the selections. Furthermore, the uncertainties due to modelling, for example through avoidance of the temporal dimension when accounting biomass-based carbon emissions to and sequestration from the atmosphere, may be of central importance. Although uncertainties may be great and the importance of including them in LCA has long been recognized (Heijungs & Huijbregts 2004), they are still often ignored in LCA studies (Finnveden et al. 2009). Similarly, most of the studies concerning differentiation of emission reduction commitments between countries (e.g. Phylipsen et al. 1998, den Elzen et al. 2005, 2006, 2007, 2008a, b, Höhne et al. 2005, 2006) have not conducted uncertainty analysis in a comprehensive manner.

In climate change mitigation, greater attention should be paid to uncertainties related to various emission reduction measures, in order to promote primarily the most certain ones. If the precautionary principle is followed, more conservative rather than optimistic estimates of emission reduction potentials of technologies should be used. The emission leakage has increased and became a serious risk to the effectiveness of climate policy and emission reductions implemented, for example, in the EU. Agreement on a comprehensive climate convention with ambitious emission reduction targets would lower the emission leakage risk significantly. An equitable solution in effort sharing is one of the major barriers to the success of international climate negotiations. If such an agreement cannot be achieved, the role of introducing consumption-based criteria and/or emission regulation at product level increases.

It is reasonable to ask whether the LCA is ready to move from an analysis tool to a decision tool such as the one applied in the context of the EU sustainability criteria for transportation biofuels and other bioliquids (RED). Applying the RED methodology to select the biofuels to be promoted in the EU cannot ensure that GHG emissions are reduced, as the consequences are not captured by the methodology. Careful consideration of market effects through resource competition should be carried out by using system level analysis. An integrated use of models with specific advantages is suggested. General and partial equilibrium models may be used to describe the interlinkages of energy and land use under the given economic conditions to generate more robust GHG emission scenarios that can be further analysed by climatic models. When the target is to reduce emissions, it is not necessarily important to model everything exactly, but to create incentives which lead to appropriate consequences.

The results of an LCA and system level top-down and bottom-up modelling will only be useful if their audience perceives the results to be relevant. Results of such analyses are increasingly applied to justify various decisions by different stakeholders such as policy-makers and consumers. As concluded by Williams et al. 2009, the future of LCA depends to a great extent on how the community decides to handle uncertainty. The same holds true for system level top-down and bottom-up modelling (Creutzig et al. 2012). Insufficient efforts puts public trust in the field at risk, and therefore transparency and handling of uncertainty related to methodological choices, parameters and modelling must be improved. Harmonisation of the practices and data management systems from goal and scope definition to interpretation phase should be systematically developed. Thus, conscious misuse of the LCA framework and system level modelling to warrant various decisions, and disinform public and private decision-makers can be avoided.

Acknowledgements

VTT Technical Research Centre of Finland, BIOGHG project of the Climbus programme and BIOVAIKU and SUBICHOE projects of the BioRefine programme of the Finnish Funding Agency for Technology and Innovation (Tekes), FOBIT project of the Academy of Finland (124421), the EU Bioenergy Network of Excellence project, the Finnish Environmental Cluster Research Programme, the Finnish Ministry of the Environment, the Ministry of Employment and the Economy (former the Ministry of Trade and Industry), Pohjolan Voima Oy, Vapo Oy and Neste Oil Oyj are gratefully acknowledged for funding.

References

Ahlgren, S., Hansson, P.-A., Kimming, M., Aronsson, P., Lundqvist, H. 2009. Greenhouse gas emissions from cultivation of agricultural crops for biofuels and production of biogas from manure. 2009-09-08, Revised version. Dnr SLU ua 12-4067/08.

Al-Riffai, P., Dimaranan, B., Laborde, D. 2010. Global Trade and Environmental Impact Study of the EU Biofuels Mandate. IFPRI.

Barona, E., Ramankutty, N., Hyman, G., Coomes, O.T. 2010. The role of pasture and soybean in deforestation of the Brazilian Amazon. Environmental Research Letters 5, 024002.

Birur, D., Hertel, T., Tyner, W. 2008. Impact of Biofuel Production on World Agricultural Markets: A Computable General Equilibrium Analysis. GTAP Working Paper No. 53.

Bringezu, S., Schütz, H., O'Brien, M., Kauppi, L., Howarth, R.W., McNeely, J. 2009. Towards sustainable production and use of resources: assessing biofuels. United Nations Environment Programme (UNEP) & International Panel for Sustainable Resource Management.

Bringezu, S., Fischer-Kowalski, M., Kleijn, R., Palm, V. (Eds.) 1997. Analysis for action: support for policy towards sustainability by material flow accounting. Proceedings of the conaccount conference 11–12 September. Wuppertal Special 6. Wuppertal Institute, Germany.

BIOGRACE 2012. Harmonised Calculations of Biofuel Greenhouse gas Emissions in Europe. Available: http://www.biograce.net.

CALCAS 2009. Co-ordination action for innovation in life-cycle analysis for sustainability. Available: http://fr1.estis.net/sites/calcas/default.asp.

Cherubini, F., Peters, G., Bernsten, T., Strømman, A., Hertwich, E. 2011. CO_2 emissions from biomass combustion for bioenergy: atmospheric decay and contribution to global warming. Global Change Biology Bioenergy 3(5), 413–426.

Chertow, M. 2000. The IPAT Equation and Its Variants. Changing Views of Technology and Environmental Impact. Journal of Industrial Ecology 4(4), 13–27.

Cornelissen, S., Dehue, B., Wonink, S. 2009. Summary of approaches to account for and monitor indirect impacts of biofuel production. PECPNL084225, Ecofys.

Creutzig, F., Popp, A., Plevin, R., Luderer, G., Minx, J., Edenhofer, O. 2012. Reconciling top-down and bottom-up modelling on future bioenergy deployment. Nature Climate Change 2, 320–327.

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.

Dansgaard, W., Johnsen, S.J., Clausen, H.B., Dahl-Jensen, D., Gundestrup, N.S., Hammer, C.U., Hvidberg, C.S., Steffensen, J.P., Sveinbjörnsdottir, A.E., Jouzel, J., Bond, G. 1993. Evidence for general instability of past climate from a 250-kyr ice-core record. Nature 364, 218–220.

Davis, S.J., Caldeira, K. 2010. Consumption-based accounting of CO_2 emissions. PNAS 107(12), 5687–5692.

DeLucchi, M.A. 1993. Greenhouse-gas emissions from the use of new fuels for transportation and electricity. Transportation Research Part A, 27A(3), 187–191.

den Elzen, M., Lucas, P., van Vuuren, D. 2005. Abatement costs of post-kyoto climate regimes. Energy Policy 33(16), 2138–2151.

den Elzen, M.G., Berk, M., Lucas, P., Criqui, P., Kitous, A. 2006. Multi-stage: a rule- based evolution of future commitments under the climate change convention. International Environmental Agreements: Politics, Law and Economics 6(1), 1–28.

den Elzen, M.G., Höhne, N., Brouns, B., Winkler, H., Ott, H.E. 2007. Differentiation of countries' future commitments in a post- 2012 climate regime: an assessment of the south–north dialogue proposal. Environmental Science & Policy 10(3), 185–203.

den Elzen, M., Höhne, N., Moltmann, S. 2008a. The triptych approach revisited: a staged sectoral approach for climate mitigation. Energy Policy 36(3), 1107–1124.

den Elzen, M.G.J., Lucas, P.L., VanVuuren, D.P. 2008b. Regional abatement action and costs under allocation schemes for emission allowances for achieving low CO_2-equivalent concentrations. Climatic Change 90(3), 243–268.

den Hond, F. 2000. Industrial ecology: a review. Regional Environmental Change 1(2), 60–69.

de Santi, G., Edwards, R., Szekeres, S., Neuwahl, F., Mahieu, V. (Eds.) 2008. Biofuels in the European context: facts and uncertainties. European Commission Joint Research Centre, JRC.

Doornbosch, R., Steenblik, R. 2007. Biofuels: is the cure worse than the disease. Round table on sustainable development. OECD, Paris.

EC 1996. Communication on Community Strategy on Climate Change. Council Conclusions. European Council, Brussels.

EC 2007. Commission of the European Communities 2007. Communication from the Commission to the Council and the European Parliament – Renewable Energy Road Map; Renewable energies in the 21st century: building a more sustainable future. COM(2006) 848 final. Brussels 10.1.2007.

EC 2008. Package of Implementation Measures for the EU's Objectives on Climate Change and Renewable Energy for 2020. European Commission, Brussels, 2008.

EC 2010a. Communication from the Commission on the practical implementation of the EU biofuels and bioliquids sustainability scheme and on counting rules for biofuels. Official Journal of the European Union. 2010/C 160/02.

EC 2010b. Communication from the Commission on voluntary schemes and default values in the EU biofuels and bioliquids sustainability scheme. Official Journal of the European Union. 2010/C 160/01.

EC 2010c. Commission decision of 10 June 2010 on guidelines for the calculation of land carbon stocks for the purpose of Annex V to Directive 2009/28/EC. Official Journal of the European Union. 2010/335/EU.

EC 2010d. Report from the Commission on indirect land-use change related to biofuels and bioliquids. 811 final. Brussels, 22.12.2010.

EC 2011a. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. Energy Efficiency Plan 2011. 109 final. Brussels, 8.3.2011.

EC 2011b. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. A Roadmap for moving to a competitive low carbon economy in 2050. 112 final. Brussels, 8.3.2011.

EC 2011c. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. Roadmap to a Resource Efficient Europe. 571 final. Brussels, 20.9.2011.

EC 2011d. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. Energy Roadmap 2050. 885/2. Brussels.

Ecometrica 2009. A Practical Approach for Policies to Address GHG Emissions from Indirect Land Use Change Associated with Biofuels. Technical Paper – TP-080212-A. Greenergy, January 2009.

Edwards, R., Mulligan, D., Marelli, L. 2010. Indirect Land Use Change from increased biofuels demand. Comparison of models and results for marginal biofuels production from different feedstocks. European Commission, Joint Research Centre, Institute for Energy. Available: http://re.jrc.ec.europa.eu/bf-tp.

Edwards, R., Griesemann, J.-C., Larivé, J.-F., Mahieu, V. 2003. Well-to-Wheels Analysis of Future Automotive Fuels and Powertrains in the European-Context. Jointly carried out by EUCAR, CONCAWE and JRC/IEA. Well-to-Tank Report Version 1, December 2003.

Ekvall, T., Tillman, A.-M., Molander, S. 2005. Normative ethics and methodology for life cycle assessment. Journal of Cleaner Production 13(13–14), 1225–1234.

Ekvall, T., Weidema, B.P. 2004. System boundaries and input data in consequential life cycle inventory analysis. The International Journal of Life Cycle Assessment 9(3), 161–171.

Elsayed, M.A., Matthews, R., Mortimer, N.D. 2003. Carbon and energy balances for a range of biofuel options. 21/3 Final Report. Sheffield Hallam University, Resources Research Unit, United Kingdom. 341 p.

EU 2009a. Directive 2009/29/EC of the European parliament and of the council of 23 April 2009 amending Directive 2003/87/EC so as to improve and extend the greenhouse gas emission allowance trading scheme of the Community. Official Journal of the European Union, 5.6.2009.

EU 2009b. Decision No 406/2009/EC of the European Parliament and of the Council of 23 April 2009 on the effort of Member States to reduce their greenhouse gas emissions to meet the Community's greenhouse gas emission reduction commitments up to 2020. Official Journal of the European Union, 5.6. 2009.

EU 2009c. 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. Official Journal of the European Union, 5.6.2009.

Eurostat 2008. Eurostat Database. Available: ec.europa.eu/eurostat.

Fabiosa, J.F., Beghin, J.C., Dong, F., Elobeid, A., Tokgoz, S., Tun-Hsiang, Y. 2010. Land Allocation Effects of the Global Ethanol Surge: Predictions from the International FAPRI Model. Land Economics 86, 687–706.

Fargione, J., Hill, J., Tilman, D., Polasky, S., Hawthorne, P. 2008. Land clearing and the biofuel carbon debt. Science 319(5867), 1235–1238.

Ferrao, P., Nhambiu, J. 2009. A comparison between conventional LCA and hybrid EIO-LCA: Analyzing crystal giftware contribution to global warming potential. In Handbook of Input-Output Economics in Industrial Ecology; Springer, New York.

Finnveden, G., Hauschild, M.Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., 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(12), 1165–1173.

Forsström, J., Pingoud, K., Pohjola, J., Valsta, L., Vilén, T., Verkerk, H. 2012. Wood-based biodiesel in Finland. Market-mediated impacts on emissions and costs. VTT Technology 7. VTT, Espoo, Finland. 47 p. + app. 1 p.

Frischknecht, R. 2000. Allocation in life cycle inventory analysis for joint production. The International Journal of Life Cycle Assessment 5(2), 85–95.

Frischknecht, R., Althaus, H.J., Bauer, C., Doka, G., Heck, T., Jungbluth, N., Kellenberger, D., Nemecek, T. 2007. The environmental relevance of capital goods in life cycle assessments of products and services. The International Journal of Life Cycle Assessment 11, 1–11.

Fritsche, U. 2007. GHG Accounting for Biofuels: Considering CO_2 from Leakage; Extended and updated version. Working paper prepared for BMU by Oeko-Institut. May 21, 2007, Darmstadt (Germany).

Gnansounou, E., Panichelli, L., Dauriat, A., Villegas, J.D. 2008. Accounting for indirect land-use changes in GHG balances of biofuels. Review of current approaches. Working paper Ref. 437.101. March 2008, Lausanne.

Graus, W., Worrel, E. 2011. Methods for calculating CO_2 intensity of power generation and consumption: a global perspective. Energy Policy 39, 613–627.

Groenenberg, H., Phylipsen, D., Blok, K. 2001. Differentiating commitments world wide: global differentiation of GHG emissions reductions based on the triptych approach – a preliminary assessment. Energy Policy 29(12), 1007–1030.

Guinée, J.B., Heijungs, R., Huppes, G. 2004. Economic allocation: examples and derived decision tree. The International Journal of Life Cycle Assessment 9(1), 23–33.

Hansen, J., Sato, M., Kharecha, P., Beerling, D., Berner, R., Masson-Delmotte, V., Pagani, M., Raymo, M., Royer, D.L., Zachos, J.C. 2008. Target Atmospheric CO_2: Where Should Humanity Aim? The Open Atmospheric Science Journal 2, 217–231.

Harding, K.G., Dennis, J.S., von Blottnitz, H., Harrison, S. 2007. A life-cycle comparison between inorganic and biological catalysis for the production of biodiesel. Journal of Cleaner Production 16, 1368–1378.

Heijungs, R., Huijbregts. M. A. J. 2004. A review of approaches to treat uncertainty in LCA. In Complexity and integrated resources management. Proceedings of the 2^{nd} biennial meeting of the International Environmental Modelling and Software Society (iEMSs), edited by C. Pahl-Wostl et al. Manno. International Environmental Modelling and Software Society, Switzerland.

Hendrickson, C.T., Lave, L.B., Matthews, H.S. 2006. Environmental Life Cycle Assessment of Goods and Services: An Input-Output Approach. RFF Press, London.

Heston, A., Summers, R., Aten, B. 2007. Penn World Table, Version 6.2. Center for International Comparisons of Production, Income and Prices at the University of Pennsylvania. Available: http://pwt.econ.upenn.edu.

Houghton, R.A. 2009. Emissions of carbon from land management. Background note for Development and Climate Change. World Development Report 2010. The World Bank, Washington, DC.

Huijbregts, M.A.J. 2001. Uncertainty and variability in environmental life-cycle assessment. PhD thesis. Institute for Biodiversity and Ecosystem Dynamics, University of Amsterdam.

Höhne, N., Phylipsen, D., Ullrich, S., Blok, K. 2005. Options for the second commitment period of the Kyoto Protocol. Climate Change 02–05, Research Report 203 41 148/01. Ecofys GmbH.

Höhne, N., Phylipsen, D., Moltmann, S. 2006. Factors underpinning future action. Technical Report, Ecofys GmbH.

IEA 2004. Energy Statistics Manual. International Energy Agency (IEA), Statistical Office of the European Communities (EUROSTAT). IEA publications 9. Paris, 2004.

IEA 2010a. Energy technology perspectives 2010. Scenarios & Strategies to 2050. International Energy Agency (IEA). Paris, 2010.

IEA 2010b. CO_2 emissions from fuel combustion database. International Energy Agency (IEA). Paris, 2010.

IEA 2010c. Energy balances database. International Energy Agency (IEA). Paris, 2010.

IEA 2010d. Electricity information. International Energy Agency (IEA). Paris, 2010.

IPCC 1996. Second Assessment Report: Climate Change 1995 (SAR). Working Group I Report "The Science of Climate Change". Available: http://www.ipcc.ch/publications_and_data/publications_and_data_reports.shtml.

IPCC 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Available: http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.htm.

IPCC 2007a. Fourth Assessment Report (AR 4). Working Group I Report "The Physical Science Basis". Available: http://www.ipcc.ch/publications_and_data/publications_and_data_reports.shtml.

IPCC 2007b. Fourth Assessment Report (AR 4). Working Group II Report "Impacts, Adaptation and Vulnerability". Available: http://www.ipcc.ch/publications_and_data/publications_and_data_reports.shtml.

IPCC 2007c. Fourth Assessment Report (AR 4). Working Group III Report "Mitigation of Climate Change". Available: http://www.ipcc.ch/publications_and_data/publications_and_data_reports.shtml.

IPCC 2009. IPCC Expert Meeting on the Science of Alternative Metrics. The Grand Hotel, Oslo, Norway 18–20 March 2009. Meeting Report. Available: http://www.ipcc.ch/pdf/supporting-material/expert-meeting-metrics-oslo.pdf.

IPCC 2011. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Prepared by Working Group III of the Intergovernmental Panel on Climate Change [O. Edenhofer, R. Pichs-Madruga, Y. Sokona, K. Seyboth, P. Matschoss, S. Kadner, T. Zwickel, P. Eickemeier, G. Hansen, S. Schlömer, C. von Stechow (Eds)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. 1075 p.

ISO 2006:14040. Environmental management – Life cycle assessment – Principles and framework. International Organization for Standardization (ISO), 2006. 20 p.

ISO 2006:14044. Environmental management – Life cycle assessment – Requirements and guidelines. International Organization for Standardization (ISO), 2006. 46 p.

JEC 2011. Well-to-wheel analysis of future automotive fuels and powertrains in the European context. Joint Research Centre, CONCAWE, EUCAR WELL-to-TANK Report. Version 3c, July 2011.

Keeney, R.L., Raiffa, H. 1993. Decisions with multiple objectives: preferences and value tradeoffs. Cambridge University Press.

Kendall, A., Chang, B., Sharpe, B. 2009. Accounting for time-dependent effects in biofuel life cycle greenhouse gas emissions calculations. Environmental Science & Technology 43(18), 7142–7147.

Kim, S., Dale, B.E. 2005. Life cycle inventory information of the United states electricity system. The International Journal of Life Cycle Assessment 10(4), 294–304.

Kim, S., Dale, B.E. 2011. Indirect land use change for biofuels: testing predictions and improving analytical methodologies. Biomass and Bioenergy 35(7), 3235–3240.

Kirkinen, J., Minkkinen, K., Sievänen, R., Penttilä, T., Alm, J., Saarnio, S., Silvan, N., Laine, J., Savolainen, I. 2007. Greenhouse impact due to different peat fuel utilisation chains – a life cycle approach. Boreal Environment Research 12, 211–223.

Kirkinen, J., Soimakallio, S., Mäkinen, T., Savolainen, I. 2010. Greenhouse impact assessment of peat-based Fischer–Tropsch diesel life-cycle. Energy Policy 38(1), 301–311.

Kirkinen, J., Palosuo, T., Holmgren, K., Savolainen, I. 2008. Greenhouse Impact Due to the Use of Combustible Fuels: Life Cycle Viewpoint and Relative Radiative Forcing Commitment. Environmental Management 42(3), 458–469.

Klaassen, G., Riahi, K. 2007. Internalizing externalities of electricity generation: an analysis with MESSAGE-MACRO. Energy Policy 35(2), 815–827.

Kline, K.L., Oladosu, G.A., Dale, V.H., Mc Bride, A.C. 2011. Scientific analysis is essential to assess biofuel policy effects: In response to the paper by Kim and Dale on “Indirect land use change for biofuels: testing predictions and improving analytical methodologies”. Biomass and Bioenergy 35, 4488–4491.

Koljonen, T., Flyktman, M., Lehtilä, A., Pahkala, K., Peltola, E., Savolainen, I. 2009. The role of CCS and renewables in tackling climate change. Energy Procedia 1(1), 4323–4330.

Kujanpää, M., Eggers, J., Verkerk, H., Helin, T., Lindner, M., Wessman, H. 2010. Carbon balance of forest residue collection and combustion in Southern-Finland. Proceedings of the 18th European Biomass Conference and Exhibition, Lyon.

Lee, K.M., Lee, S.Y., Hur, T. 2004. Life cycle inventory analysis for electricity in Korea. Energy 29(1), 87–101.

Lifset, R., Graedel, T. 2002. In Ayres, R.U., Ayres, L.W. (Eds.). A Handbook of Industrial Ecology. Edward Elgar, Cheltenham.

Lloyd, S.M., Ries, R. 2007. Characterizing, Propagating, and Analyzing Uncertainty in Life-Cycle Assessment. A Survey of Quantitative Approaches. Journal of Industrial Ecology 11(1), 161–179.

Loulou, R. 2008. ETSAP-TIAM: the TIMES integrated assessment model part II: mathematical formulation. Computational Management Science 5(1–2), 41–66.

Loulou, R., Labriet, M. 2008. ETSAP-TIAM: the TIMES integrated assessment model part I: model structure. Computational Management Science 5(1–2), 7–40.

Loulou, R., Remme, U., Kanudia, A., Lehtilä, A., Goldstein, G. 2005. Documentation for the times model. Technical Report DM 70046/ICC03080, IEA Energy Technology Systems Analysis Programme (ETSAP). Available: http://www.etsap.org/documentation.asp.

Lund, H., Mathiesen, B.V., Christensen, P., Schmidt, J.H. 2010. Energy system analysis of marginal electricity supply in consequential LCA. The International Journal of Life Cycle Assessment 15(3), 260–271.

Malca, J., Freire, F. 2011. Life-cycle studies of biodiesel in Europe: A review addressing the variability of results and modeling issues. Renewable and Sustainable Energy Reviews 15, 338–351.

Manne, A., Mendelsohn, R., Richels, R. 1995. Merge a model for evaluating regional and global effects of GHG reduction policies. Energy Policy 23(1), 17–34.

Mathiesen, B.V., Munster, M., Fruergaard, T. 2009. Uncertainties related to the identification of the marginal energy technology in consequential life cycle assessments. Journal of Cleaner Production 17(15), 1331–1338.

Mattila, T.J., Pakarinen, S., Sokka, L. 2010. Quantifying the Total Environmental Impacts of an Industrial Symbiosis – a Comparison of Process-, Hybrid and Input-Output Life Cycle Assessment. Environmental Science & Technology 44(11), 4309–4314.

Mattila, T., Leskinen, P., Soimakallio, S., Sironen, S. 2012. Uncertainty in environmentally conscious decision making: beer or wine? The International Journal of Life Cycle Assessment 17(6), 696–705.

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B. 2007. Key elements in framework for land use impact assessment within LCA. The International Journal of Life Cycle Assessment 12(1), 5–15.

Mitchell, D.O. 2008. A note on rising food prices. Policy Research Working paper 4682, Development Prospects Group. World Bank, Washington, DC, 2008.

Mongelli, I., Suh, S., Huppes, G. 2005. A Structure Comparison of two approaches to LCA inventory data, based on the MIET and ETH Databases. The International Journal of Life Cycle Assessment 10(5), 317–324.

Mäkinen, T., Soimakallio, S., Paappanen, T., Pahkala, K., Mikkola, H. 2006. Greenhouse gas balances and new business opportunities for biomass-based transportation fuels and agrobiomass in Finland. Technical Research Centre of Finland. VTT Research Notes 2357. VTT, Espoo, Finland. 134 p.+ app.19 p. (In Finnish).

Nijkamp, P., Wang, S., Kremers, H. 2005. Modeling the impacts of international climate change policies in a CGE context: the use of the GTAP-E model. Economic Modelling 22(6), 955–974.

NOAA 2012. Trends in Atmospheric Carbon Dioxide. The National Oceanic and Atmospheric Administration Earth System Research Laboratory. Available: http://www.esrl.noaa.gov/gmd/ccgg/trends/index.html#global.

Nordhaus, W.D. 1999. Roll the DICE again: the economics of global warming. Version rice 98 pap 121898.wpd. Yale University; January 28, 1999.

O'Hare, M., Delucchi, M., Edwards, R., Fritsche, U., Gibbs, H., Hertel, T., Hill, J., Kammen, D., Laborde, D., Marelli, L., Mulligan, D., Plevin, R., Tyner, W. 2011. Comment on "Indirect land use change for biofuels: testing predictions and improving analytical methodologies" by Kim and Dale: statistical reliability and the definition of the indirect land use change (iLUC) issue. Biomass and Bioenergy 35, 4485–4487.

Overmars, K.P., Stehfest, E., Ros, J.P.M., Gerdien Prins, A. 2011. Indirect land use change emissions related to EU biofuel consumption: an analysis based on historical data. Environmental science and policy 14, 248–257.

Peltonen-Sainio, P., Jauhiainen, L., Hannukkala, A. 2007. Declining rapeseed yields in Finland: how, why and what next? Journal of Agricultural Science 145, 587–598.

Peltonen-Sainio, P., Jauhiainen, L. 2010. Cultivar improvement and environmental variability in yield removed nitrogen of spring cereals and rapeseed in northern growing conditions according to a long-term dataset. Agricultural and food science 19, 341–353.

Peters, G.P., Marland, G., Le Quéré, C., Boden, T., Canadell, J.G., Raupach, M.R. 2012. Rapid growth in CO_2 emissions after the 2008–2009 global financial crisis. Nature climate change 2, 2–4.

Peters, G., Minx, J., Weber, C.L, Edenhofer, O. 2011. Growth in emission transfers via international trade from 1990 to 2008. PNAS 108(21), 8903–8908.

Peters, G.P., Marland, G., Hertwich, E.G., Saikku, L., Rautiainen, A., Kauppi, P. 2009. Trade, transport, and sinks extend the carbon dioxide responsibility of countries: An editorial essay. Climatic Change 97(3–4), 379–388.

Peters, G., Hertwich, E. 2008. CO_2 Embodied in International Trade with Implications for Global Climate Policy. Environmental Science & Technology 42(5), 1401–1407.

Petit, J.R., Jouzel, J., Raynaud, D., Barkov, N. I., Barnola, J.-M., Basile, I., Bender, M., Chappellaz, J., Davisk, M., Delaygue, G., Delmotte, M., Kotlyakov, V.M., Legrand, M., Lipenkov, V.Y., Lorius, C., Pépin, L., Ritz, C., Saltzmank, E., Stievenard, M. 1999. Climate and atmospheric history of the past 420,000 years from the Vostok ice core, Antarctica. Nature 399, 429–436.

Phylipsen, D., Höhne, N., Janzic, R. 2004. Implementing triptych 6.0 – technical report. Technical Report.

Phylipsen, G.J.M., Bode, J.W., Blok, K., Merkus, H., Metz, B. 1998. A triptych sectoral approach to burden differentiation; ghg emissions in the European bubble. Energy Policy 26(12), 929–943.

Plevin, R.J., O'Hare, M., Jones, A.D., Torn, M.S., Gibbs, H.K. 2010. Greenhouse gas emissions from biofuels indirect land use change are uncertain but may be much greater than previously estimated. Environmental Science & Technology 44(21), 8015–8021.

Pingoud, K., Cowie, A., Bird, N., Gustavsson, L., Rüter, S., Sathre, R., Soimakallio, S., Türk, A., Woess-Gallasch, S. 2010. Bioenergy: Counting on Incentives. Letter. Science 5, 1199–1200.

Pingoud, K., Ekholm, T., Savolainen, I. 2011. Global warming potential factors and warming payback time as climate indicators of forest biomass use. Mitigation and Adaptation Strategies for Global Change 17(4), 369–386.

Ray, A. 1984. Cost Benefit Analysis: Issues and Methodologies. Johns Hopkins University Press, Baltimore, MD.

Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Schmidt, W.-P., Suh, S., Weidema, B.P., Pennington, D.W. 2004. Life cycle assessment – Part 1: Framework, goal & scope definition, inventory analysis, and applications. Environment International 30(5), 701–720.

Reijnders, L., Huijbregts, M.A.J. 2008. Palm oil and the emission of carbon-based greenhouse gases. Journal of Cleaner Production 16(4), 477–482.

Reinhardt, G. 1991. Biofuels: Energy and GHG balances: Methodology and Case Study Rape Seed Biodiesel. Institut für Energie- und Umweltforschung, Heidelberg, Germany.

Repo, A., Tuomi, M., Liski, J. 2011. Indirect carbon dioxide emissions from producing bioenergy from forest harvest residues. Global Change Biology Bioenergy 3(2), 107–115.

Righelato, R., Spraclen, D.V. 2007. Carbon mitigation by biofuels or by saving and restoring forests. Science 317(5840), 902.

Ringius, L., Torvanger, A., Holtsmark, B. 1998. Can multi-criteria rules fairly distribute climate burdens? OECD results from three burden sharing rules. Energy Policy 26(10), 777–793.

Rioual, P., Andrieu-Ponel, V., Rietti-Shati, M., Batterbee, R.W., de Beaulieu, J.-L., Cheddadi, R., Reille, M., Svobodova, H., Shemesh, A. 2001. High-resolution record of climate stability in France during the last interglacial period. Nature 413, 293–296.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hugher, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U. Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J.A. 2009. A safe operating space for humanity. Nature 461, 472–475.

Runge, C.F., Senauer, B. 2007. How biofuels could starve the poor. Foreign Affairs 86(3), 41–53.

Saaty, T.L. 1980. The Analytic Hierarchy Process: planning, priority setting, resource allocation. McGraw-Hill, New York.

Santoyo-Castelazo, E., Gujpa, H., Azapagic, A. 2011. Life cycle assessment of electricity generation in Mexico. Structural analysis of electricity consumption by productive sectors. The Spanish Case Energy 36(3), 1488–1499.

Saikku, L., Rautiainen, A., Kauppi, P.E. 2008. The sustainability challenge of meeting carbon dioxide targets in Europe by 2020. Energy Policy 36(2), 730–742.

Saikku, L., Soimakallio, S., Pingoud, K. 2012. Attributing land-use change carbon emissions to exported biomass. Environmental Impact Assessment Review 37, 47–54.

Sathaye, J., Norgaard, R., Makundi, W. 1993. A conceptual Framework for the Evaluation of Cost-Effectiveness of Projects to Reduce GHG Emissions and Sequester Carbon. LBL-33859, Lawrence Berkeley Laboratory, Berkeley, CA, USA.

Schlamadinger, B., Edwards, R., Byrne, K.A., Cowie, A., Faaij, A., Green, C., Fijan-Parlov, S., Gustavsson, L., Hatton, T., Heding, N., Kwant, K., Pingoud, K., Ringer, M., Robertson, K., Solberg, B., Soimakallio, S., Woess-Gallasch, S. 2005. Optimizing the greenhouse gas benefits of bioenergy systems. 14th European Biomass Conference, 17–21 October 2005, Paris, France, 4p.

Scott-Wilson 2009. Developing a spreadsheet model for the calculation of the emissions from indirect land use change (ILUC) as a result of biofuel production. Explanatory note, prepared for FoE UK. London, 2009.

Searchinger, T., Heimlich, R., Houghton, R.A., Dong, F., Elobeid, A., Fabiosa, J., Tokgoz, S., Hayes, D., Yu, T.-H. 2008. Use of U.S. Croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 319, 1238–1240.

Socolow, R., Andrews, C., Berkhout, F., Thomas, V. (Eds.) 1994. Industrial Ecology and Global Change. Cambridge University Press, UK.

Soimakallio, S., Antikainen, R., Thun, R. (Eds.) 2009. Assessing the sustainability of liquid biofuels from evolving technologies – A Finnish approach. VTT Research Notes 2482.

Soimakallio, S., Perrels, A., Honkatukia, J., Moltmann, S., Höhne, N. 2006. Analysis and Evaluation of the Triptych 6 – Case Finland. VTT Working Papers 48. VTT, Espoo, Finland. Available: http://www.vtt.fi/inf/pdf/workingpapers/2006/W48.pdf.

Squire, L., van der Tak, H. 1975. Economic Analysis of Projects. Johns Hopkins University Press, Baltimore, MD, USA.

Statistics Finland 2006. Greenhouse gas emissions in Finland 1990–2004. National Inventory Report to the European Union, 15 January 2006.

Suh, S. 2004. Functions, commodities and environmental impacts in an ecological-economic model. Ecological Economics 48(4), 451–467.

Suh, S., Lenzen, M., Treloar, G.J., Hondo, H., Horvath, A., Huppes, G., Jolliet, O., Klann, U., Krewitt, W., Moriguchi, Y., Munksgaard, J., Norris, G. 2004. System boundary selection in lifecycle inventories using hybrid approaches. Environmental Science & Technology 38(3), 657–664.

Suh, S., Huppes, G. 2005. Methods for Life Cycle Inventory of a product. Journal of Cleaner Production 13(7), 687–697.

Syri, S., Lehtila, A., Ekholm, T., Savolainen, I., Holttinen, H., Peltola, E. 2008. Global energy and emissions scenarios for effective climate change mitigation – deterministic and stochastic scenarios with the TIAM model. International Journal of Greenhouse Gas Control 2(2), 274–285.

Tol, R. 2003. Is the Uncertainty about Climate Change too Large for Expected Cost-Benefit Analysis? Climatic Change 56(3), 265–289.

Torvanger, A., Ringius, L. 2001. Burden Differentiation: Criteria for Evaluation and Development of Burden Sharing Rules. CICERO Center for International Climate and Environmental Research, Oslo, Norway.

Tsupari, E., Monni, S., Pipatti, R. 2005. Non-CO_2 greenhouse gas emissions from boilers and industrial processes. Evaluation and update of emission factors for the Finnish national greenhouse gas inventory. VTT Research Notes 2321. VTT, Espoo, Finland. 82 p. + app. 24 p.

Tsupari, E., Monni, S., Tormonen, K., Pellikka, T., Syri, S. 2007. Estimation of annual CH_4 and N_2O emissions from fluidised bed combustion: an advanced measurement-based method and its application to Finland. International Journal of Greenhouse Gas Control 1(3), 289–297.

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

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 (WIA-2). The International Journal of Life Cycle Assessment 4, 66–74 and 4, 167–174.

Udo de Haes, H.A., Finnveden, G., Goedkoop, M., Hauschild, M., Hertwich, E.G., Hofstetter, P., Jolliet, O., Klöpffer, W., Krewitt, W., Lindeijer, E.W., Müller-Wenk, R., Olsen, S.I., Pennington, D.W., Potting, J., Steen, B. (Eds.) 2002. Life-Cycle Impact Assessment: Striving Towards Best Practise. SETAC Press, Pensacola, FL.

UNEP 2012. United Nations Environment Programme. Available: http://www.unep.org.

UNFCCC 1992. United Nations Framework Convention on Climate Change. United Nations 1992. Available: http://unfccc.int/resource/docs/convkp/conveng.pdf.

UNFCCC 2010. Report of the Conference of the Parties on its sixteenth session, held in Cancun from 29 November to 10 December 2010. FCCC/CP/2010/7/Add.1. Addendum, Part Two: Action taken by the Conference of the Parties at its sixteenth session, 15 March 2011. Available: http://unfccc.int/resource/docs/2010/cop16/eng/07a01.pdf#page=2.

UNFCCC 2011a. Outcome of the work of the Ad Hoc Working Group on Long-term Cooperative Action under the Convention. Draft decision [-/CP.17]. Available: http://unfccc.int/files/meetings/durban_nov_2011/decisions/application/pdf/cop17_lcaoutcome.pdf.

UNFCCC 2011b. Greenhouse gas inventory data. United Nations Framework Convention on Climate Change (UNFCCC). Available: http://unfccc.int/ghg_data/items/3800.php.

van Vuuren, D.P., den Elzen, M.G.J., Lucas, P.L., Eickhout, B., Strengers, B.J., van Ruijven, B., Wonink, S., van Houdt, R. 2007. Stabilizing greenhouse gas concentrations at low levels: an assessment of reduction strategies and costs. Climatic Change 81(2), 119–159.

van Vuuren, D.P., Hoogwijk, M., Barker, T., Riahi, K., Boeters, S., Chateau, J., Scrieciu, S., van Vliet, J., Masui, T., Blok, K., Blomen, E. 2009. Comparison of top-down and bottom-up estimates of sectoral and regional greenhouse gas emission reduction potentials. Energy Policy 37(12), 5125–5139.

VTT 2006. LIPASTO calculation system for traffic exhaust emissions and energy consumption in Finland. Technical Research Centre of Finland. Available: http://lipasto.vtt.fi/indexe.htm.

WCED 1987. Our common future. World Commission on Environment and Development. Oxford: Oxford University Press, 1987, p. 43.

Weber, C.L., Jaramillo, P., Marriott, J., Samaras, C. 2010. Life cycle assessment and grid electricity: what do we know and what can we know? Environmental Science & Technology 44, 1895–1901.

Weisser, D. 2007. A guide to life-cycle greenhouse gas (GHG) emissions from electric supply technologies. Energy 32, 1543–1559.

Weyant, J. 2000. An introduction to the economics of climate change policies. PEW center, Arlington, VA, USA.

Wihersaari, M. 2005. Aspects on bioenergy as a technical measure to reduce energy related greenhouse gas emissions. VTT Publications 564. VTT, Espoo, Finland. 93 p. + app. 71 p.

Williams, E.D., Weber, C.L., Hawkins, T.R. 2009. Managing Uncertainty in Life Cycle Inventories. Journal of Industrial Ecology 13(6), 928–944.

Appendix A: Supporting Data for Paper IV

Table A1. The annual production-based CO_2 emission intensity of electricity (g CO_2/kWh_e) in various OECD countries. The CO_2 emissions from combined heat and power production (CHP) allocated to power and heat on the basis of *the energy content of the products*. NA = data not available.

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
Australia	970	964	1035	1068	1117	1100	1077	1094	1141	1118	1070
Austria	248	212	185	217	209	276	265	259	254	234	210
Belgium	372	392	330	316	312	313	324	317	304	293	293
Canada	236	212	258	268	254	259	239	227	216	243	210
Czech	867	892	808	790	740	668	662	655	653	697	674
Denmark	680	559	405	397	388	413	347	316	396	363	351
Finland	213	246	192	239	259	332	281	177	281	253	185
France	126	86	86	68	74	80	77	90	80	86	79
Germany	724	687	608	627	646	600	584	561	554	608	547
Greece	1240	1173	1012	1049	1004	966	972	976	900	923	899
Hungary	662	657	696	659	630	722	658	545	491	475	449
Iceland	0	2	0	0	0	0	0	0	0	1	1
Ireland	887	861	753	773	742	682	674	668	614	572	544
Italy	680	644	589	572	605	615	515	507	493	473	476
Japan	484	460	446	447	470	494	475	474	465	502	488
Korea	581	611	603	592	490	476	506	493	495	485	494
Luxembourg	NA	NA	NA	NA	437	429	398	408	419	416	420
Mexico	658	663	715	729	732	748	691	753	705	713	566
Netherlands	656	592	506	523	525	533	506	494	483	485	469
New Zealand	149	131	268	332	302	346	317	365	324	286	239
Norway	1	2	1	3	2	3	3	2	3	4	3
Poland	1071	1100	1011	998	999	997	988	984	926	936	902
Portugal	622	675	558	517	600	481	540	604	488	440	455
Slovak Republic	449	377	266	290	258	316	280	274	268	270	249
Spain	511	550	519	451	524	450	453	470	403	437	367
Sweden	11	16	14	15	20	26	20	18	20	15	15
Switzerland	13	14	15	14	14	14	15	17	16	15	15
Turkey	716	671	711	741	629	574	531	544	552	608	631
United Kingdom	813	631	560	574	555	578	578	574	598	589	572
United States	705	690	685	694	654	654	659	659	639	634	622
EU-27	560	510	462	457	466	465	444	438	433	446	417
OECD Total	579	553	543	550	537	537	531	531	521	528	507

Table A2. The annual production-based CO_2 emission intensity of electricity (g CO_2/kWh_e) in various countries. The CO_2 emissions from combined heat and power production (CHP) allocated fully to power (*the "motivation electricity" method*). NA = data not available.

	1990	**1995**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**
Australia	975	964	1035	1068	1117	1100	1077	1094	1141	1118	1070
Austria	305	270	229	268	256	331	321	311	314	288	262
Belgium	407	422	345	333	327	335	349	336	336	318	319
Canada	238	217	264	275	261	267	246	233	222	249	216
Czech	1209	1290	1070	1068	995	903	895	896	873	898	889
Denmark	1065	912	722	727	703	705	643	620	674	653	663
Finland	330	369	310	366	393	468	400	313	417	388	316
France	126	89	103	89	96	102	99	117	105	113	104
Germany	818	748	663	670	688	674	676	648	632	659	601
Greece	1240	1173	1018	1056	1011	976	981	987	912	931	908
Hungary	860	807	815	758	716	832	749	624	629	588	554
Iceland	0	2	0	0	0	0	0	0	0	1	1
Ireland	887	861	753	773	742	682	674	668	614	572	544
Italy	680	644	589	572	605	615	580	574	559	538	545
Japan	484	460	446	447	470	494	475	474	465	502	488
Korea	581	622	621	655	551	533	580	561	562	554	560
Luxembourg	NA	NA	NA	NA	475	478	440	459	471	463	477
Mexico	658	663	715	729	732	748	691	753	705	713	566
Netherlands	696	743	685	700	696	707	682	651	631	622	602
New Zealand	149	131	268	332	302	346	317	365	350	310	245
Norway	2	3	3	4	3	5	6	4	5	6	4
Poland	1819	1588	1389	1393	1385	1387	1372	1358	1268	1271	1229
Portugal	629	685	575	535	620	503	566	634	516	468	485
Slovak Republic	676	574	366	369	313	404	354	341	328	346	310
Spain	512	550	519	451	524	450	453	470	403	437	367
Sweden	40	61	51	51	66	79	66	57	65	52	53
Switzerland	25	26	26	25	26	27	28	32	31	28	28
Turkey	716	671	726	753	644	586	543	564	574	631	652
United Kingdom	813	631	560	574	555	578	578	574	598	589	572
United States	709	706	696	705	666	666	668	669	653	648	636
EU-27	672	593	526	520	527	533	519	512	504	510	481
OECD Total	612	585	572	579	566	571	565	563	554	558	536

Table A3. The annual consumption-based CO_2 emission intensity of electricity in OECD countries that trade electricity over the country borders (g CO_2/kWh_e). The CO_2 emissions from combined heat and power production (CHP) allocated to power and heat on the basis of *the energy content of the products*. NA = data not available.

	1990	**1995**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**
Austria	332	291	317	337	340	395	366	374	368	381	328
Belgium	374	377	310	268	288	297	313	311	291	287	294
Canada	256	220	272	283	267	279	258	244	236	258	230
Czech	899	894	823	810	773	726	715	718	698	732	696
Denmark	398	492	309	361	337	418	318	208	383	280	242
Finland	201	240	185	241	255	339	295	199	294	285	245
France	133	89	87	71	77	85	80	94	85	93	84
Germany	696	650	581	597	612	572	557	532	530	579	525
Greece	1223	1159	1004	1032	979	949	946	943	875	906	880
Hungary	719	647	572	550	511	588	559	466	448	424	419
Ireland	887	861	751	773	737	676	667	660	613	573	545
Italy	582	550	505	485	511	523	453	441	434	409	420
Luxembourg	NA	NA	NA	NA	569	525	522	512	451	507	486
Mexico	658	664	715	729	732	748	691	753	705	713	566
Netherlands	628	507	495	518	496	512	512	499	489	496	465
Norway	1	5	3	14	11	24	18	5	14	9	6
Poland	1017	1080	995	970	974	977	965	957	909	903	867
Portugal	613	664	553	511	590	477	523	576	473	440	435
Slovakia	489	445	428	434	426	487	454	448	446	541	433
Spain	508	532	500	439	505	440	444	460	396	429	362
Sweden	12	19	19	25	36	86	59	29	50	26	23
Switzerland	178	143	161	157	187	195	171	246	196	220	182
Turkey	714	671	707	734	628	575	534	546	554	610	632
United Kingdom	783	601	540	558	542	571	563	558	583	576	555
USA	701	683	679	689	650	651	655	654	634	629	616

Table A4. The annual consumption-based CO_2 emission intensity of electricity in OECD countries that trade electricity over the country borders (g CO_2/kWh_e). The CO_2 emissions from combined heat and power production (CHP) allocated fully to power (*the "motivation electricity" method*). NA = data not available.

	1990	**1995**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**
Austria	413	365	385	412	410	479	448	455	450	452	396
Belgium	408	412	332	282	311	326	346	341	330	320	330
Canada	258	225	278	290	274	286	265	250	243	264	237
Czech	1306	1292	1097	1101	1045	986	971	981	935	952	919
Denmark	630	803	549	616	573	661	552	405	616	487	444
Finland	347	394	324	409	428	536	472	381	479	456	412
France	133	92	105	91	99	107	102	120	110	119	109
Germany	789	714	641	647	661	649	653	625	612	635	585
Greece	1228	1166	1014	1047	995	969	967	967	898	927	903
Hungary	858	798	682	641	585	682	642	551	560	520	512
Ireland	887	861	751	773	737	676	667	660	613	573	545
Italy	584	551	509	489	516	528	513	502	494	468	484
Luxembourg	NA	NA	NA	NA	604	584	599	586	512	550	533
Mexico	658	664	715	729	732	748	691	753	705	713	566
Netherlands	669	636	641	664	637	660	668	637	617	614	577
Norway	2	10	7	27	21	44	36	10	26	16	11
Poland	1688	1548	1362	1353	1348	1356	1336	1320	1243	1218	1174
Portugal	621	672	569	527	608	496	544	600	495	462	458
Slovakia	656	646	581	571	552	648	602	592	581	705	564
Spain	509	533	502	440	506	441	445	461	397	429	363
Sweden	39	63	56	67	88	161	119	71	107	67	65
Switzerland	203	163	184	178	210	227	206	293	234	251	213
Turkey	715	671	726	752	648	589	546	567	577	633	653
United Kingdom	783	601	540	558	542	571	564	559	584	577	556
USA	705	699	690	700	662	663	664	664	649	643	629

PAPER I

Greenhouse gas balances of transportation biofuels, electricity and heat generation in Finland

Dealing with the uncertainties

In: Energy Policy 37(1), 80–90.

Energy Policy 37 (2009) 80–90

Contents lists available at ScienceDirect

Energy Policy

journal homepage: www.elsevier.com/locate/enpol

Greenhouse gas balances of transportation biofuels, electricity and heat generation in Finland—Dealing with the uncertainties

S. Soimakallio[a,*], T. Mäkinen[a], T. Ekholm[a], K. Pahkala[b], H. Mikkola[b,1], T. Paappanen[a]

[a] VTT Technical Research Centre of Finland, P.O. Box 1000, FI-02044 VTT, Espoo, Finland
[b] MTT Agrifood Research Finland, FI-31600 Jokioinen, Finland

ARTICLE INFO

Article history:
Received 15 January 2008
Accepted 11 August 2008
Available online 19 September 2008

Keywords:
Emission reduction
Greenhouse gases (GHG)
Liquid biofuels

ABSTRACT

One way to reduce greenhouse gas emissions from the transportation sector is to replace fossil fuels by biofuels. However, production of biofuels also generates greenhouse gas emissions. Energy and greenhouse gas balances of transportation biofuels suitable for large-scale production in Finland have been assessed in this paper. In addition, the use of raw materials in electricity and/or heat production has been considered. The overall auxiliary energy input per energy content of fuel in biofuel production was 3–5-fold compared to that of fossil fuels. The results indicated that greenhouse gas emissions from the production and use of barley-based ethanol or biodiesel from turnip rape are very probably higher compared to fossil fuels. Second generation biofuels produced using forestry residues or reed canary grass as raw materials seem to be more favourable in reducing greenhouse gas emissions. However, the use of raw materials in electricity and/or heat production is even more favourable. Significant uncertainties are involved in the results mainly due to the uncertainty of N_2O emissions from fertilisation and emissions from the production of the electricity consumed or replaced.

1. Introduction

The promotion of renewable energy has a central position in the European Union's energy and environmental policy. The underlying target is to lower greenhouse gas emissions and mitigate the climate change. Targets have been set for different sectors, such as primary energy use, consumption of electricity and transportation fuels. The transportation sector covers a significant and further growing share of greenhouse gas emissions and it is highly dependent on fossil oil.

The Directive 2003/30/EC of the European Parliament and of the Council (EC, 2003) aims to promote the use of biofuels or other renewable fuels to replace diesel or petrol for transport purposes in each Member State, with a view to contribute objectives such as meeting the climate change commitments, securing an environmentally friendly energy supply and promoting the renewable energy sources. The Directive sets an indicative national target of 5.75% of all the petrol and diesel for road transport purposes placed on their markets by 2010 calculated on the basis of energy content.

In the "Green Paper—Towards a European Strategy for the Security of Energy Supply" (EC, 2000) the European Commission states that the European Union is consuming more and more energy and importing more and more energy products. The Community production is insufficient for the Union's energy requirements. As a result, the external dependence for energy is constantly increasing; in the next 20–30 years, if no measures are taken, from 50% to 70%. In the "Green Paper—A European Strategy for Sustainable, Competitive and Secure Energy" from the year 2006 the European Commission outlines suggestions and alternative courses of actions for the European energy future. It states that the European countries are well behind their target of 5.75% of transportation fuels.

The Commission of the European Union sets three main targets in its Strategy for Biofuels (EC, 2006). These targets are the promotion of biofuel in the EU and developing countries as well, the preparation for the large-scale use of biofuels by enhancing their competitiveness and increasing the research on second generation biofuels and the support of developing countries with a potential for economic growth from sustainable biomass production.

In the Communication from the Commission "An Energy Policy for Europe" (EC, 2007a) as part of the Strategic European Energy Review together with the Communication from the Commission "Renewable Energy Road Map" (EC, 2007b) and the Communication from the Commission "Biofuels Progress Report" (EC, 2007c), the European Commission proposes to set a binding minimum target for biofuels of 10% of vehicle fuel by 2020 and to ensure that the biofuels used are sustainable in nature, inside and outside the

* Corresponding author. Tel.: +358 20 722 6767; fax: +358 20 722 7001.
E-mail address: sampo.soimakallio@vtt.fi (S. Soimakallio).
[1] Currently at: University of Helsinki, Koetilantie 3, Finland.

0301-4215/$ - see front matter
doi:10.1016/j.enpol.2008.08.003

EU. The main drivers for this target are the mitigation of climate change, the reduction of oil-dependency and the improvement of security of supply. The particular target together with the demand to reduce greenhouse gas emissions by at least 35% compared to reference fuels to be replaced was published as a part of an integrated proposal for climate action of the European Commission (EC, 2008a–c). In addition, the demand for more comprehensive sustainability criteria was claimed.

At the end of 2005, the Ministry of Trade and Industry of Finland set up a task force to assess the promotion of biofuel use and production in the transport sector in Finland. The task force suggested that an obligation law should form the primary tool to promote the use of biofuels. On 9 June 2006, the working group set by the Ministry of Trade and Industry of Finland on climate and energy issues defined an indicative target of 5.75% for 2010. The Finnish Parliament passed the obligation law in February 2007, and it will be in force starting from the beginning of 2008. The biofuel shares set in the new Law are 2%, 4% and 5.75% for 2008, 2009 and 2010, respectively.

One of the means to achieve reductions in greenhouse gas emissions of transportation is to increase the use of biomass-based fuels. However, the determination of the potential to reduce greenhouse gas emissions is not a straightforward process as it heavily depends on the approach selected and the assumptions made in defining the system boundary. In addition, different raw materials and process options vary significantly in terms of auxiliary energy requirement and greenhouse gas emissions caused (see e.g., Edwards et al., 2003, 2008). Various recent studies have recognised changes in soil carbon balances, nitrous oxide emissions from soils and indirect land-use changes as major sources of uncertainty influencing greenhouse gas balances of biofuels (see e.g., Edwards et al., 2008; Crutzen et al., 2007; Searchinger et al., 2008; Fargione et al., 2008). That is why identifying auxiliary energy consumption and the environmental impact over the whole life cycle of biofuels is of importance when considering reductions in greenhouse gas emissions by replacing fossil fuels.

Finland is one of the northernmost countries in the world with a relatively cold climate and a short growing season. Carbon dioxide emissions from transportation have contributed to some 15–20% of the overall greenhouse gas emissions without emissions from land-use changes and forestry in Finland in recent years (Statistics Finland, 2008).

The objective of this paper was to assess energy and greenhouse gas balances for biomass-based fuels used in transportation as well as the greenhouse gas impact when reference fuels are replaced. Both commercial technologies and technologies under development were studied. Most suitable technologies for large-scale production in Finnish conditions were taken into consideration. For comparison, the greenhouse gas impact of using certain raw materials also in electricity and/or heat production was considered. Particular attention was paid to the variation of the results due to uncertainties and sensitivities involved. Consequently, the results provide a more general rather than case-specific picture of the issue.

2. Materials and methods

2.1. Approach

2.1.1. Reference land use

When assessing reductions in greenhouse gas emissions at least two conditions have to be confronted and compared to each other. The results depend fundamentally on the selections of conditions in comparisons and on system boundaries. The key question is what happens if the particular bioenergy chain is implemented or not. The reference land use and system boundaries should be defined by carefully responding to this question.

Bioenergy production chains may be implemented in fields already used for cultivation, uncultivated lands or in forests. In addition, different types of waste components may be used as raw materials. The possible impact of implementing a certain bioenergy chain on land use has to be taken into account when setting reference land use and system boundaries. If waste streams are used, no impact on land use typically takes place.

Here, the reference land use for agrobiomass-based bioenergy chains is assumed to be a set-aside as it would be the most likely option for bioenergy production in the short term. Therefore, the calculation of auxiliary energy inputs and emissions for agrobiomass chains begins from cultivation.

Forest residues are harvested after logging timber for industrial purposes. The current share of forest residues utilised in Finland (approximately 2 TWh) is small compared to the techno-economic potential corresponding to some 24 TWh (Electrowatt-Ekono, 2005). As forest industrial activities are not dependent on forest residue utilisation, no reference land use was assumed for forest-residue-based bioenergy chains. An assumption was made that more forest residues will be available than will be utilised for the studied bioenergy production chains. Therefore, if forest residues are not harvested they are left to decay, which is the basis for calculating auxiliary energy inputs and emissions.

2.1.2. System boundaries and allocation

Defining system boundaries and selection of the reference case are one of the most crucial phases of energy and greenhouse gas balance analysis. Allocation of energy inputs and emissions is a third important issue. These definitions have a significant impact on results and should be carefully considered.

There is no unique and unambiguous way for allocation procedures, and allocation should be avoided whenever possible. In this paper, allocation of different products was avoided by extending the system boundaries to also cover the use of products and by using the substitution method. Energy inputs and emissions as well as credits from the substitution of other products by co-generated products are then allocated to considered biofuels. This substitution method is often the most appropriate method.

Protein animal meal generated in the ethanol or biodiesel process was assumed to replace the use of soy protein imported from the USA. The credits in energy inputs and emission outputs of such a substitution are calculated in accordance with Edwards et al. (2003). Turnip rape-based glycerine produced in biodiesel (RME) production was assumed to be used for energy in heat production boilers to replace peat. Straw was not assumed to be harvested.

Certain issues are uniformly excluded in all fuel chains considered. For example, the energy input required and the emissions output caused by the construction of infrastructure, the production of facilities, machinery or other equipment required in overall fuel production chains were not considered. This exclusion was necessary as reliable data of such energy inputs and emissions outputs are not available. An assumption was made that the difference in the above-mentioned issues is not significant between biofuel and fossil fuel chains. In addition, the impact on emissions from ash recycling and biomass storing was not considered.

The functional unit for a transportation fuel was assumed to be one kilometre driven and for electricity (and heat) production one kilowatt hour produced. Emission reductions were calculated by replacing reference fuels with the biofuels considered.

2.1.3. Assessment of the greenhouse impact

Calculations for energy inputs and greenhouse gas emissions were carried out by following as uniform principles as possible for all the chains. Energy inputs were converted into primary energy by using certain factors depending on the form of energy required. Energy inputs include both fossil and renewable energy required in the fuel production chains, but not the energy which is transferred into fuel itself (i.e. only auxiliary energy inputs expressed as primary energy were considered).

Carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) were considered when calculating greenhouse gas emissions. The greenhouse effect was studied by using the approach known as *Global Warming Potential* (GWP), and the emissions of CH_4 and N_2O were converted to carbon dioxide equivalents by using the time frame of 100 years and emission factors of 21 and 310, respectively (IPCC, 1996a).

According to Revised 1996 IPCC Guidelines (IPCC, 1996b), CO_2 emissions from biomass combustion are regarded as recycling atmospheric CO_2 if biomass is extracted from a sustainable (i.e. replenished) source. These CO_2 emissions are therefore excluded from net emission calculations.

2.2. Parameters for biofuel chains

2.2.1. Electricity

Electricity consumed within fuel production chains was assumed to be purchased from the electric grid. The Finnish electricity grid is connected to the Nordic electricity system containing various electricity production forms varying from very low emission (e.g. hydro power) to high emission (e.g. coal condensing power) sources. When aiming to mitigate climate change, the conservation of energy has been argued as one of the main emission reduction options by the EU (EC, 2008d). At the same time, there are plans to increase electricity consumption, e.g. by increasing the production of biofuels (EC, 2008c).

There is no generally accepted consensus on how the emissions from electricity production should be evaluated. These issues are discussed, e.g. by Curran et al. (2005). It is very typical that the greenhouse gas impact of electricity consumption for a certain purpose is assumed in accordance with the average production mix of a market area. The overall electricity consumption of a market area consists of numerous single consumption points and production forms at the same time. In principle, it is reasonable to consider that no consumption point can have just a certain kind of production but an average mix for all. However, this kind of approach is in conflict with the fundamental principle of the reference surface defined as it does not consider system impacts objectively.

When assessing the greenhouse gas impact of any electricity consuming process the reference surface should be considered as not implementing the particular process. If not implemented, the particular amount of electricity is conserved. In other words, the greenhouse gas impact of electricity consumption should be analogically valued as equal to the impact of conservation of that same amount of electricity. Decrease or increase in electricity consumption has a direct impact on the marginal side of electricity production constructed typically by high emission electricity production forms (see e.g., Kara et al., 2008; Holttinen and Tuhkanen 2004). Therefore, by valuing consumed electricity as the average mix of a market area, the impact on the greenhouse gas emissions of electricity conservation is underestimated.

It is also possible that the electricity required at the consumption point will increase the use of renewable energy sources that would have not otherwise taken place. For example, a consumer may generate the required electricity himself by using renewable energy sources or may buy some green certificates to ensure that the production of renewable energy sources will be increased by the amount consumed by the consumer. In these cases, it would be reasonable to assume that the electricity consumption would equal the particular production form instead of marginal electricity. However, if the particular consumption point reduces the amount of low emission electricity sold in the grid to replace marginal electricity, the electricity consumption should again be handled as marginal electricity.

According to the methodology defined above, in general, electricity consumed at a particular consumption point can be assumed to correspond to marginal electricity, some particular electricity production form (e.g. wind power or bioenergy) or anything in between. In addition, marginal electricity varies from year to year depending mainly on consumption, the availability of hydropower and the prices of fuels and emission allowances in the EU emission trading scheme (EU ETS). It is likely that marginal production will gradually move towards lower emission intensity, when tighter emission reductions will be required, but the change can be slow.

As a consequence of the very random nature of electricity consumed at certain general consumption points, a wide range of 0–900 g CO_2-eq./kWh_e was defined for emissions. The upper limit corresponds to typical current marginal electricity production in an average precipitation year in the Nordic electricity markets simulated by VTT (e.g. Kara et al., 2008). In addition, Holttinen and Tuhkanen (2004) concluded that wind power connected to the electric grid in the Nordic countries replaces mainly coal-fired power generation which is analogical with the given upper limit mentioned above.

The primary energy demand of electricity production was assessed according to calculation methods of Statistics Finland (2005) and by assuming that 10% additional primary energy is required to produce fuels used in electricity production. The primary energy factor for electricity consumption used was 2.35 kWh_{prim}/kWh_e with a variation of $\pm 20\%$.

2.2.2. Fossil fuels

The primary energy demand and the greenhouse gas emissions of diesel fuel, heavy fuel oil and natural gas required in machinery and equipment of fuel production chains were estimated in accordance with Edwards et al. (2003). As CH_4 and N_2O emissions from fuel combustion depend heavily on combustion technology and conditions, emissions for each type of machinery or boilers was assessed individually based on Statistics Finland (2006) and VTT (2006). Similarly, the specific fuel consumption and the greenhouse gas emissions of transport were estimated for each type of truck and assumed loading factors by using the LIPASTO (2006) calculation system of VTT (2006).

2.2.3. Fertilisers

The production of fertilisers, particularly nitrogen fertilisers, is energy intensive. In addition, nitrous oxide emissions during the production of nitric acid may be significant. Therefore, the energy input and the greenhouse gas emissions of fertilisers depend fundamentally on the nitrogen content. The primary energy input and the greenhouse gas emissions from fertiliser production (Table 1) were assessed in accordance with Kemira fertilisers (H. Hero, personal communication, October 2005).

Nitrous oxide emissions are also generated due to nitrification and denitrification processes caused by micro-organism activity in soil. Part of the nitrogen content of the fertiliser is converted directly to N_2O and part of it indirectly through nitrogen oxides (NO_x) and ammonium (NH_3) (Monni et al., 2007). The amount of these emissions is uncertain but may be significant. By using the

Table 1
Primary energy input and emissions from production and transportation of fertilisers as well as emissions from fertliser use (H. Hero, personal communication, October 2005; IPCC, 2006; Statistics Finland, 2006)

Fertiliser (Kemira)	Production and transportation				N_2O from soil			Total		
	Primary energy	Emissions (kg CO_2-eq./t)			Emissions (kg CO_2-eq./t)			Emissions (kg CO_2-eq./t)		
	GJ/t	Min	Def.	Max	Min	Def.	Max	Min	Def.	Max
Syysviljan Y1	7.8	636	736	938	163	1028	4507	798	1764	5444
Kevätviljan Y2	13.1	1097	1276	1632	288	1819	7973	1385	3095	9605
Kevätviljan Y3	11.9	977	1132	1442	250	1582	6933	1227	2713	8375

default values of IPCC (2006) and uncertainty boundaries given by Statistics Finland (2006) used for national greenhouse gas inventories it was assumed that the direct and indirect N_2O emissions from agricultural soils together are equal to 1.6235 kg N_2O-N/kg N-fertiliser and vary from 0.2566 to 7.116 kg N_2O-N/kg N-fertiliser. Crutzen et al. (2007) assessed that the global average of N_2O emissions from N fertilisers vary between some 3 and 5 kg N_2O-N/kg N fertiliser. The range given by Crutzen et al. (2007) is significantly higher compared to the default values given by IPCC (2006) but lower compared to the upper limit set in this paper based on field measurements and expert judgement (Monni et al., 2007).

Fertilisation of forest lands after removing forest residues may be necessary to compensate for nutrient loss due to outage of nitrogen. The loss may be compensated by nitrogen deposit or ash circulation. Wihersaari (2005) estimated that if compensation fertilisation of nitrogen would take place it would cause emissions of 7 kg CO_2-eq./MWh_{chip}. This was set as an upper limit for forest land fertilisation emissions considered in this paper with the lower limit of 0 kg CO_2-eq./MWh and the default value determined by given Weibull distribution (Table A1).

2.2.4. *Limestone and pesticides*

Different types of carbonate compounds are used to reduce the acidity of agricultural soil. Carbonate of limestone then reacts in soil and emits carbon dioxide into the atmosphere. The amount of carbon dioxide generated depends on the soil properties and the carbonate compound. The average emission factor for the use of limestone was defined according to Statistics Finland (2006) by taking into account the use of different types of carbonate compounds in Finland between 1990 and 2004, to equal 431 kg CO_2/t varying from 388 to 474 kg CO_2/t. In addition, the primary energy demand and the greenhouse gas emissions to produce and transport limestone were assessed to equal 0.5 GJ/t and 21 kg CO_2-eq./t, respectively.

According to Edwards et al. (2003), the primary energy input and the emissions from pesticide production vary significantly in different literature sources. However, the amount of pesticides used is relatively small, and therefore the impact on overall emissions is low. Default values given by Edwards et al. (2003) were used and equalled 267 GJ/t and 16 666 kg CO_2-eq./t. The range was from 175 to 516 GJ/t and from 10 642 to 31 379 kg CO_2-eq./t, respectively.

2.2.5. *Soil carbon balances*

A part of the carbon content of biomass is sequestered into soil if biomass is not harvested. Therefore, biomass harvesting decreases carbon content in soil. Part of the stored carbon may be released as methane, which is reduced by harvesting logging residues. However, as the amount of these emissions is not well-known, the possible compensating impact of the harvesting procedure is not considered in this paper.

If logging residues are harvested, soil carbon pools are decreased by 1–2% during the rotation of one tree generation (Palosuo et al., 2001). According to Yasso model calculations (Liski et al., 2005) approximately 2–10% of the original carbon content of logging residues would have been stored in soils during the first 100 years after felling, if not harvested. As this carbon content is released into the atmosphere during combustion, it can be seen as an indirect carbon emission over 100 years resulting from the harvesting and combustion of logging residues. Wihersaari (2005) estimated that this indirect emission compared to the energy content of logging residues may correspond to some 40–45 kg CO_2/MWh. In this paper, it was assumed that the upper value illustrates the upper boundary for indirect carbon emission, whereas the lower boundary is zero. The default value was set in the middle of the range.

When pristine land or virgin forest is converted to arable land, typically some amount of carbon stored in soils is released as carbon dioxide into the atmosphere. In addition, permanent harvesting and tillage result in a decline of soil carbon balance. This lowering will continue until a new balance between carbon input and output in the soil is achieved. Reduction or overall renunciation of tillage activity has been found to turn the soil carbon balance back into growth (e.g. Schjønning et al., 2007; Gregorich et al., 2005; Mikhailova et al., 2000) until a new balance between carbon input and output is achieved. Accumulation of soil carbon in mineral soils depends on the rate at which organic matter is added to the soil and the rate at which erosion and biological oxidation remove the organic matter from the soil (Reicosky et al., 1995). Similarly, the cultivation of viable perennial crops, such as reed canary grass, requires infrequent ploughing and therefore increases overall soil carbon stock until a balance is achieved. However, the quantitative changes in soil carbon balances are not well-known.

By using very rough factors given by IPCC (2006) intended for greenhouse gas inventories, it was assumed that the upper limit for annual change in soil carbon balance during 100 years is equal to -0.078 and 0.003 t C/ha for conventional tillage cultivation and the no-tillage option or cultivation of reed canary grass, respectively. The default values were set in the middle of the range.

2.2.6. *Fuel processing chemicals*

Small amounts of chemicals, for example sulphuric and phosphoric acid, smectite, caustic soda and hexane, are required and consumed in biofuel processing. Figures presented by Elsayed et al. (2003) were used for the energy input and emissions of the production and transportation of these chemicals.

2.3. Biofuel chains

2.3.1. Raw material production

Six different barley cultivation chains and five turnip rape chains were assessed. The production chains were formed by varying the method of soil tillage, the method of the seeding (seeding with a standard combined seed drill after tillage or direct drilling), the method of grain storage (hot air drying or storage of moist grains in an airtight silo) and the energy source for grain drying (oil or wood chips). The yield level of direct drilling was assumed to be 10% lower compared with the yield of the standard combined seed drill after tillage. Cultivation chains and the transportation of barley and turnip rape are presented in more detail by Mikkola and Pahkala (2008) and Mäkinen et al. (2006).

Reed canary grass represents the most yielding field energy plant in Finland, with a typical annual yield level varying from 4.5 to 8 t_{dm}/ha depending on soil type and fertilisation. Reed canary grass cultivation also represents a possible reuse option for decommissioned peatlands. According to Mäkinen et al. (2006), reed canary grass was assumed to be cultivated in set-aside agricultural lands and harvested either as loose material or baled bundles.

The harvesting, transportation and chipping of logging residues was assessed for three different commercial and established options: chipping at the end-use facility by harvesting as loose residue or as bundles or chipping at the roadside.

Cultivation or harvesting and transportation chains are presented with default values in Mäkinen et al. (2006). Those default figures with defined uncertainty ranges for the main parameters are presented in Appendix A. Ranges selected for yield levels represent the natural variation in Finland due to changes in climatic conditions and do not therefore correlate with the range assumed for fertilisation rates, which only indicate human errors.

2.3.2. Fuel processing, distribution and combustion

All liquid biofuels considered were assumed to be processed in industrial scale plants with an annual capacity of 28–47 and 79–105 kilotons biofuel produced for the technologies commercial and under development, respectively. Synthetic biofuel production (here F-T diesel) was assumed to be integrated into a modern pulp and paper mill or a paper mill, which provides significant advantages in overall energy efficiency.

The mass and energy balances of RME processing were assumed in accordance with Elsayed et al. (2003); for other fuels the expertise of the Technical Research Centre of Finland was used in setting default values (Mäkinen et al., 2006). The mass and energy balances with assumed uncertainty ranges are presented in Appendix B.

2.4. Reference chains

2.4.1. Diesel oil and gasoline

Fossil diesel oil and gasoline were used as reference fuels for liquid biofuels. In addition, diesel oil is required in the production of biofuels. For auxiliary energy requirements and greenhouse gas emissions from diesel oil and gasoline production chain (well-to-tank) figures given by Edwards et al. (2003) were used (Table 2).

Table 2
Energy input and greenhouse gas emissions from well to tank for diesel oil and gasoline (Edwards et al., 2003)

Fuel	Primary energy (MJ/kg)			Greenhouse gas emissions (g CO_2-eq./MJ)		
	Min	Def.	Max	Min	Def.	Max
Diesel oil	0.14	0.16	0.18	12.6	13.8	16.0
Gasoline	0.12	0.14	0.17	11.1	11.7	14.6

2.4.2. Replaced electricity and/or heat

For the comparison of the greenhouse gas impact of replacing fossil fuels in transportation the use of reed canary grass and logging residues in power and/or heat production was estimated. Direct fuel switching, e.g. replacing coal or peat by bioenergy in proportion to the effective heating value of the fuels in existing power and/or heating plants is a relatively straightforward way to estimate the greenhouse gas impact. However, fuel switching in existing power and/or heating plants is not always appropriate and not the only possible option to be considered.

Difficult methodological problems are encountered when considering new electricity and/or heating capacity based on bioenergy replacing something else. First, problems occur similar to those discussed in Section 2.2.1 to estimate the greenhouse gas emissions of marginal electricity. Second, there is not necessarily load for produced heat, which restricts the possibility to consider objectively CHP or stand-alone heating plants. Third, the production rates for heat and power may vary between technologies causing further methodological problems as the reference entity should be equal for the technologies in comparison (see e.g., Gustavsson and Karlsson, 2002).

The most probable options for defining reference fuels are direct fuel switching in existing power and/or heating plants and the replacement of certain fuels in foreseen plants, as well as the replacement of marginal electricity in stand-alone power plants. All these choices with independent uncertainties increase the sensitivity and stochastic nature of the greenhouse gas impact considered. However, the replacement of marginal electricity with its minimum and maximum values will very likely provide boundaries for calculating the credits of all above-mentioned choices.

The lower limit for marginal electricity was defined as 300 g CO_2-eq./kWh to correspond to possible foreseen marginal electricity in the future (gas-fired condensing power), whereas the upper limit equals 900 g CO_2-eq./kWh in accordance with Section 2.2.1. These values were selected to illustrate the variation in greenhouse gas emissions of replaced fuels, regardless of whether bioenergy is used in current or foreseen power and/or heating plants. It should be noted that the uncertainty is significantly lower if certain technology was replaced. Consequently, applying the above-mentioned range very likely exaggerates the overall uncertainty of replacing certain technology, but is more suitable to illustrate the likely variation of a range of technologies.

2.4.3. Substitution of soybean meal and peat

Soybean meal is the main protein-rich animal feed in the EU and most of it is imported from the USA. Protein fodder co-generated in the production of ethanol from barley or biodiesel from turnip rape can be used to some extent instead of soybean meal. As the protein content of those meals is not, however, as high as in soy bean meal, more barley or turnip rape-based animal meal has to be used to replace soy bean meal. 1 kg of soy bean meal was assumed to be replaced by 1.30 and 1.28 kg of turnip rape and barley meal, respectively.

Values for credits in primary energy input (2.7 kWh/kg) and greenhouse gas emissions (230 g CO_2-eq./kg) from the substitution of soybean meal were assumed in accordance with Edwards et al. (2003). The uncertainty for both energy and greenhouse gas emission credits was assumed to be $\pm 30\%$.

Glycerine co-produced in RME biodiesel production was assumed to be used as energy in heating boilers, as the market for glycerine as a chemical is very limited. Peat was assumed as the reference fuel to be replaced in proportion to the effective heating value of the fuels. According to Kirkinen et al. (2007) and by using the GWP method, the greenhouse impact from peat production in forestry-drained peatland, peat combustion and the afforestation option during 100 years equals 107 ± 12 g CO_2-eq./MJ. Those values for replaced peat were used in calculating credits for RME from glycerine use (assumed lower heating value (LHV), 16 MJ/kg).

2.5. Uncertainty analysis

The uncertainty analysis was conducted using the Monte Carlo method. In this method a suitable probability distribution is defined for each uncertain variable, and a large number of samples, in this case 15 000, are drawn from each distribution. For each set of samples from all distributions, the desired result variables are calculated, thus giving a set of frequency distributions for the result variables. The resulting frequency distributions represent the total effect of the variability of the assumptions on the result variable, and can be taken as empirical probability density distributions through normalising their area to a value of one.

Apart from one, all of the variables were represented with a three-parameter Weibull distribution due to its flexibility. Depending on the parameterisation, the distribution can vary from symmetric to skewed, thus giving the distribution the ability to imitate a large variety of probability distributions. An exclusion to this methodology was emissions for electricity, for which a uniform distribution was assigned. The variables were chosen so that there would be no correlations between them in the model.

The contribution of a single variable to the uncertainty of a result variable was measured using Spearman's rank correlations ρ between each of the uncertainty variables and the result values. The rank correlation has the advantage over common Pearson correlation in that it does not require the dependence between the quantities to be linear but only monotonic, which happens to be the case with all parameters and results in this study.

3. Results

3.1. Energy balances

Auxiliary energy is required in every step of biofuel chains from well to wheel. The difference in various cultivation or harvesting chains of particular raw materials was only minor. The exception was direct seeding in addition to air tight storage of grains resulting in approximately 25% lower energy consumption compared to other studied options for barley production (Mäkinen et al., 2006).

The major part of auxiliary energy is consumed within fuel processing, particularly ethanol and F-T diesel. The benefit from lower energy consumption in direct drilling of barley and turnip rape was almost compensated by the drawback from lower substitution credit due to a lower amount of animal meal produced compared to conventional tillage. Consequently, the variation between different cultivation chains in overall energy consumption per energy content of fuel produced was only minor.

Energy requirements were highest for the production of barley-based-ethanol and F-T diesel in the stand-alone concept (Fig. 1). These production chains consumed almost equally auxiliary energy as what is transferred to fuel itself. The lowest energy requirement was calculated for the RME and integrated F-T diesel concept minimising purchased electricity. The ranges were the widest for ethanol and RME, mainly due to uncertainties in credits from replacing soybean meal and in yield rates. All biofuel chains considered required significantly, approximately 3–5 times, more auxiliary energy compared to fossil diesel and gasoline chains from well to tank.

3.2. Greenhouse gas balances

Greenhouse gas balances of biofuel chains may vary significantly depending on many factors, e.g. raw material, possible land-use change effects, fertilisation, yield rates, auxiliary energy requirements and the sources of energy. As with auxiliary energy inputs, the difference in greenhouse gas balances in various cultivation or harvesting chains of particular raw material was also only minor (Mäkinen et al., 2006).

In order to enable comparisons of the impact of fuels on greenhouse gas emissions the use of the fuels should be taken into account. One kilometre driven was used as a reference entity on which the greenhouse gas impact was calculated. All biofuels were considered to be used in fraction of 5 vol% mixed to gasoline or diesel. The relative impact on greenhouse gas emissions when

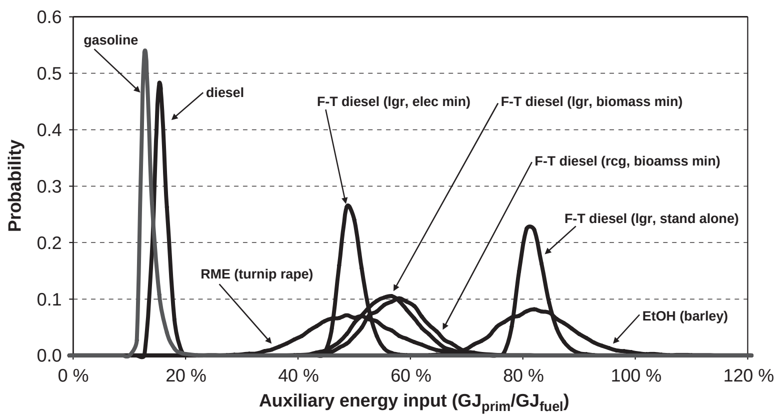


Fig. 1. Probability distributions for auxiliary energy input per fuel energy content for considered biofuels (lgr = logging residues, rcg = reed canary grass).

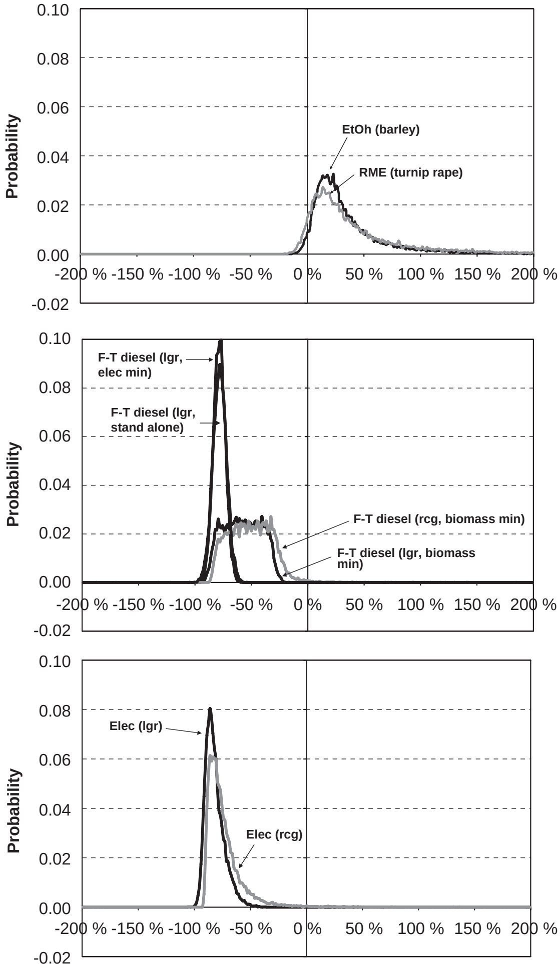


Fig. 2. (a)–(c) Probability distributions for the relative greenhouse gas impact when replacing reference fuels. For (a) and (b) Elec = electricity production, lgr = logging residues, rcg = reed canary grass. For (b) Elec min and biomass min refer to integrated F-T diesel processing cases with minimum purchased electricity and biomass, respectively.

substituting reference fuels was then calculated and it is illustrated in Fig. 2.

There is only a very low probability that greenhouse gas emissions from barley-based ethanol and turnip rape-based RME would be lower than those of gasoline and diesel (Fig. 2a). Instead, there is a moderate probability to get even 50–100% higher values. The wide uncertainty range, and particularly the high upper limit for ethanol from barley and RME from turnip rape, results mainly from a significant uncertainty in N_2O emissions from fertilisation. Yield rate, animal feed output and emissions from electricity production were the next dominant factors in the case of ethanol and RME.

Greenhouse gas emissions from producing F-T diesel depend significantly on the concept considered (Fig. 2b). If the biomass requirement is minimised, greenhouse gas emissions are highly dependent on emissions from production of electricity consumed in the process. For such concepts, the expected value of the relative greenhouse gas emission impact when replacing fossil diesel varies between −30% and −80%, depending mainly on emissions from electricity production. If the purchased electricity requirement is minimised and replaced by using more biomass, the uncertainty range is decreased significantly and the expected value for relative greenhouse gas emission impact when replacing fossil diesel is around −80%. Soil carbon losses and N_2O emissions from fertilisation added to the yield rate were the next dominant factors in the case of F-T diesel using logging residues and reed canary grass as raw materials, respectively.

The relative emission impact is probably higher when using logging residues or reed canary grass in electricity production to replace marginal electricity instead of producing biofuels of them to replace fossil diesel (Fig. 2b and c). It should be noted that the emission factor given for electricity has the opposite impact on the results in the case of replacing marginal electricity compared to electricity consumed in the case of transportation biofuels. Consequently, increase in the particular emission factor increases

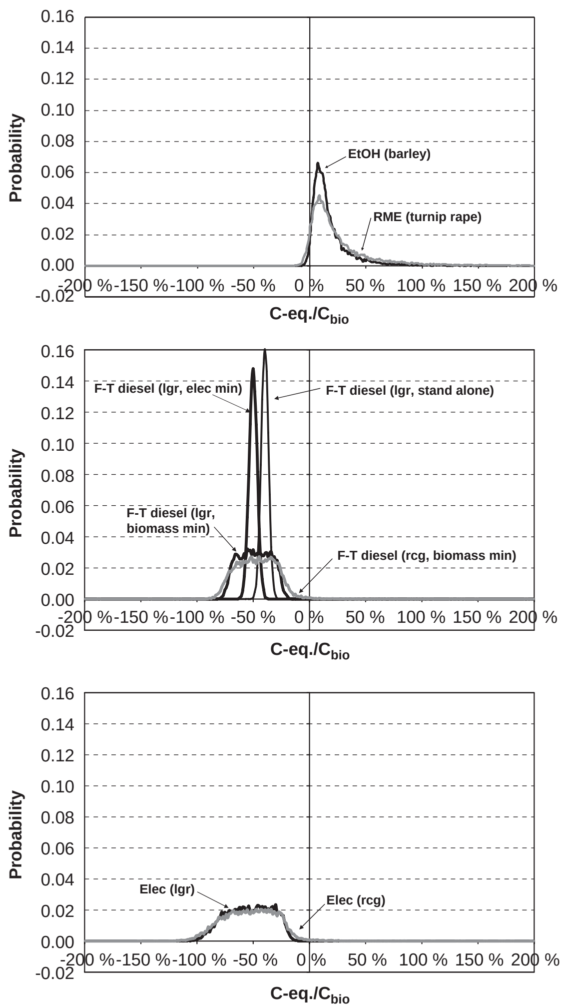


Fig. 3. (a)–(c) Probability distributions for reduced carbon equivalent emission per consumed biocarbon when replacing reference fuels. For (a) and (b) Elec = electricity production, lgr = logging residues, rcg = reed canary grass. For (b) Elec min and biomass min refer to integrated F-T diesel processing cases with minimum purchased electricity and biomass, respectively.

the relative emission impact of biofuels whereas decreases it of electricity production.

The relative greenhouse gas impact indicator gets the lower value from the higher the emissions from reference fuel replaced are, and from the lower amount of the biofuels produced, respectively (Fig. 2a–c). However, it does not indicate the effectiveness that a particular bioenergy chain has in reducing greenhouse gas emissions. In other words, a certain biofuel chain may look very favourable in terms of relative greenhouse gas emission impact, but at the same time consuming significant amount of biomass resulting in relatively low emission reduction in absolute terms. Consequently, it is necessary to consider how much greenhouse gas emissions can be reduced by consuming a certain amount of biomass (Fig. 3a–c). The most significant changes in Fig. 3 compared to Fig. 2 are the location of distributions for F-T diesel concepts based on a stand-alone solution and minimum purchased electricity use, as well as the form of probability distributions for electricity production from logging residues and reed canary grass. These changes are due to the relatively significant biomass requirement in the above-mentioned F-T diesel concepts and the more sensitive nature of this particular indicator compared to the relative emission impact against emission savings from replaced electricity.

If availability of land area is a limiting factor, it is important to consider greenhouse gas emission reduction that can be achieved from certain land areas producing biomass. This may be the case for agricultural biofuels as cultivated land area may be used for other purposes as well. This kind of indicator was not considered in this paper as it is not very well suitable for forest residue chains with no additional land requirement. However, according to the results presented in Fig. 3 it can be argued that the cultivation of reed canary grass for energy enables significantly more reductions in greenhouse gas emissions per land area than the cultivation of turnip rape or barley for energy. Similarly, more reductions per land area are very likely achieved by producing electricity and/or heat than F-T diesel from reed canary grass.

Nitrous oxide emissions, emissions from electricity production and emission savings from replaced electricity are the most significant parameters for ethanol and RME, F-T diesels, and electricity production, respectively. The uncertainties in other individual parameters have a clearly lower influence on the overall uncertainty of studied cases, but were all together not negligible (Table 3).

4. Conclusions and discussion

This paper assessed energy and greenhouse gas balances of biomass-based transportation fuels assumed to be the most suitable for large-scale production in Finland. Technologies that are both commercial and under development were considered. The results were compared to replaceable fossil fuels and also to the use of raw materials in electricity and/or heat production.

The auxiliary energy requirement in biofuel chains considered was approximately 3–5-fold compared to fossil reference fuel chains. However, all studied chains reduced the overall consumption of fossil fuels in substitution.

Significant uncertainties and sensitivities to parameter value selections are involved in the analysis, particularly concerning greenhouse gas emissions. Consequently, it was impossible to provide even nearly exact results on greenhouse gas emissions of biofuel production or on reductions in greenhouse gas emissions when replacing fossil reference fuels.

The results indicated that greenhouse gas emissions from the production and use of barley-based ethanol or biodiesel from turnip rape are very likely to be higher compared to the fossil reference fuels. The use of fertilisers is significant compared to the energy content of the barley and turnip rape yield in Finland mainly due to climatic conditions. The production and use of nitrogen fertilisers causes emissions of nitrous oxide, which are probably significant and may be very significant. The impact of various cultivation chains of barley and turnip rape on greenhouse gas emissions of ethanol and RME was only minor compared to the other uncertainties involved.

The cultivation of uncultivated or set-aside lands to produce barley-based ethanol or biodiesel from turnip rape will very likely increase the absolute emissions of greenhouse gases, regardless of the replacement of fossil fuels by the produced biofuels. Greenhouse gas emissions in absolute terms can to some extent be reduced by optimising cultivation to avoid surplus production.

Table 3
Impact of the 10 most important parameters on emission impact per biocarbon consumed for studied biofuel chains expressed as rank correlation

Parameter	EtOH	RME	F-T (lg, biom min)	F-T (lg, elec min)	F-T (lg, stand alone)	F-T (rcg, biom min)	Elec (lg)	Elec (rcg)
Emission from electricity production	0.27	0.07	**0.97**	0.36	0.09	**0.89**		
Electricity demand			0.06	0.02	0.01	0.04		
Yield rate of raw material	−0.26	−0.27				−0.16		−0.13
Carbon content in DM of raw material	−0.07					0.15	−0.01	0.12
LHV in DM of raw material			−0.04	−0.18	−0.15	−0.04	−0.04	−0.03
N_2O from soil (fertilization)	**0.84**	**0.88**	0.04	0.14	0.01	0.25	0.03	0.20
Fertiliser use	0.12	0.09				0.03		0.02
Emissions from fertiliser production	0.10	0.11						0.02
Ploughing								−0.02
Animal feed output		0.15						
Soil carbon losses	0.16	0.14	0.21	**0.84**	**0.94**		0.13	
Emission savings from replaced electricity							**−0.95**	**−0.89**
Efficiency of biofuelled power plant							−0.27	−0.26
Emissions of biofuelled power plant							0.02	
Output of produced fuel		−0.15	−0.05	−0.21	−0.19	−0.03		
Emission savings from replaced soybean meal	−0.06	−0.06						
Emissions from replaced reference fuel	−0.05		−0.06	−0.17	−0.15	−0.05		
Emissions from transportation			0.02	0.06	0.08	0.03	0.02	0.03
Emissions from forest haulage			0.01	0.02	0.02		0.01	
Emissions from chipping			0.01	0.01	0.02		−0.01	
CO_2 from liming	0.05							
Lime use		0.06						

A positive value indicates that an increase in the value of a particular parameter increases the value of the result. A negative value indicates the opposite effect. The most significant correlations (r values above 0.5) are highlighted.

The achievable emission reduction would probably be higher by reducing overproduction than by producing ethanol or biodiesel from barley and turnip rape, respectively. However, utilisation of straw to substitute fossil fuels and measures to increase the soil carbon balance and reduce nitrous oxide emissions could considerably decrease greenhouse gas emissions of the cereal crop chains. These options were not quantitatively considered in this paper due to the lack of practical solutions and information.

Second generation biofuels produced by using forestry residues or reed canary grass as raw materials seem to be more favourable in reducing greenhouse gas emissions. Lower emissions are mainly due to the significantly lower fertilisation demand per energy content of the particular raw materials compared to the cereal crops. However, the amount of biomass and purchased electricity used together with the large variation of greenhouse gas emissions from electricity production has a significant impact on the results. Consequently, the greenhouse gas impact per biocarbon consumed for F-T diesels is very likely between −30% and −70%.

The achievable emission reductions are likely higher by using the raw materials in electricity and/or heat production compared to producing liquid biofuels. This is true at least as long as there is a load for heat production or the greenhouse gas emissions from marginal electricity production which correspond approximately to the current level.

The relative greenhouse gas impact when replacing reference fuel related to greenhouse gas emissions from reference fuel does not objectively measure the effectiveness of biomass in reducing greenhouse gas emissions. Consequently, the indicator that takes into account the amount of biomass used or the area required to achieve a certain reduction in greenhouse gas emissions is more suitable for such purposes. These kinds of indicators are also suggested, e.g. by Pingoud et al. (2006) and Schlamadinger et al. (2005).

Several studies concerning greenhouse gas emissions of biofuels have been done in many contexts all over the world. The results may vary significantly between the studies due to many different reasons. First, methodological differences may occur in system boundaries, allocation procedures, as well as in reference land use, system or entity. Consideration of macro-effects such as land-use changes due to competition of raw materials is crucial but difficult to quantify and was excluded in this analysis. Figures for individual parameters may vary due to the above-mentioned issues. In addition, there may be a natural variation in parameters set due to differences in climatic conditions, cultivation or harvesting practices, fuel production processes and energy sources used, for example. All these factors make the objective comparison between different studies difficult without careful and detailed consideration of the causes for the results.

As the uncertainties associated with the greenhouse gas emissions of biofuels were found to be significant it is very difficult to define general default values that would be reliable and suitable for certain types of biofuels produced in various conditions. Therefore, the case-specific consideration of biofuels is required and cannot be excluded in policy-making. Attention should also be paid to research work aiming to reduce the uncertainty and significance of factors having remarkable impact on the results, such as nitrous oxide emissions from soils and soil carbon balances. However, the uncertainty involved in these processes may be very difficult to be reduced significantly due to lack of information, resulting in the need to accept the uncertainty associated with greenhouse gas emissions. In addition, system impacts on electricity production or land use caused by biofuel production should carefully be analysed as the implications on greenhouse gas emissions may be very significant. Consideration of these issues is crucial in the aim to prepare sustainable criteria for biofuels to avoid unwanted climatic implications.

Optimising the use of biomass is manifold and complicated as many vital and difficult issues other than mitigation of climate change are also involved. Consequently, the connection between energy-related and non-energy-related biomass cannot be cut. In addition, issues like employment, sufficient food supply, self-sufficiency and security in energy supply, famine and environmental issues other than those related to climate change are also of central importance. Typically, optimisation of one factor would not give an optimal solution for the others. Therefore, choices and compromise solutions have a significant impact on biomass production and utilisation, and furthermore on greenhouse gas emissions.

Acknowledgements

The financial support provided by Tekes, the Finnish Funding Agency for Technology and Innovation, the Ministry of Trade and Industry, the Technical Research Centre of Finland (VTT), MTT Agrifood Research Finland, Pohjolan Voima Oy, Vapo Oy and Neste Oil Oyj are gratefully acknowledged.

Appendix A

Mass and energy balances with the given uncertainty ranges for raw material cultivation or harvesting and transportation (see Table A1).

Table A1

	Barley			Turnip rape			Reed canary grass			Logging residues		
	Min	Def	Max	Min	Def	Max	Min	Def	Max	Min	Def	Max
Yield level (t/ha, a)	2.975	3.5	4.025	1.36	1.6	1.84	4	8	10		51	
Moisture content (%)	12	13	14	8	9	10	18	20	22	39	43	47
LHV in dry matter (MJ/kg_{dm})	17.1	17.4	17.8	25.9	26.4	26.9	17.2	17.6	18.0	19.0	19.5	20.0
Start up fertilization at annual level (kg/ha, a)							27.0	30.0	33.0			
Start-up fertiliser N-P-K							13-7-13	13-7-13	13-7-13			
Annual fertilization (kg/ha, a)	360	400	440	392	435	479	150	300	325			
Annual fertiliser N-P-K (%)	20-3-8	20-3-8	20-3-8	23-3-5	23-3-5	23-3-5	20-3-8	20-3-8	20-3-8			
Emissions from forest fertilization (kg CO_2-eq./MWh)										0	1.90	7
Liming at annual level (t/ha, a)	720	800	880	720	800	880	720	800	880			
Pesticides (l/ha, a)	1.04	1.3	3.96	1.76	2.2	5.04						
Transportation distance (km)	50	100	150	35	70	105	35	70	105	30	60	120
Diesel fuel consumption in cultivation/harvesting (l/ha)	109	127	146	96	112	130	56	87	119	140	164	187
Diesel fuel consumption in transportation (l/MJ, km)[a]	1.4	1.5	1.7	1.4	1.5	1.7	6.6	7.4	8.1	4.0	4.4	4.9
Electricity consumption (kWh/ha)	57	63	70	29	32	36				110	122	135

[a] Uncertainty ranges given here do not take into account factors having an impact on overall energy content of raw materials.

Table B1

Product	EtOH	RME	F-T diesel (INT 1)	F-T diesel (INT 2)	F-T diesel (SA)	F-T diesel (INT 1)
Raw material	Barley	Turnip rape	Logging residues			Reed canary grass
Annual raw material input (t/a)	183 600	75 600	481 200	499 000	791 100	243 700
Raw material moisture (%)	13 (±10)	9 (±11)	45 (±10)	45 (±10)	45 (±10)	20 (±10)
Auxiliary energy demand						
Electricity (GWh/a)	19.7 (±2%)	4.1 (±5%)	247.6 (±5%)	24.0 (±5%)	7.8 (±5%)	191.7 (±5%)
Natural gas (TJ/a)	543.4 (±2%)	88.4 (±5%)				
Heavy fuel oil (TJ/a)		4.8 (±5%)				
Diesel oil (TJ/a)		8.6 (±5%)				
Co-produced animal meal (moisture 10%) (t/a)	36 100 (±2%)	52 960 (±5%)				
Co-produced animal meal (moisture 67%) (t/a)	42 200 (±2%)					
Co-produced glycerine (t/a)		3360 (±5%)				
Methanol demand (t/a)		3360	1030	810	1030	770
LHV of the fuel product (MJ/kg)	26.8	37.5	44.0	44.0	44.0	44.0
Annual fuel production (t/a)	47 400 (±2%)	27 600 (±5%)	103 100 (±5%)	81 200 (±5%)	103 100 (±5%)	76 900 (±5%)

INT1 = unit integrated into pulp and paper mill with minimised biomass demand.
INT2 = unit integrated into pulp and paper mill with minimised electricity demand.
SA = stand-alone unit.

Appendix B

Mass and energy balances with the given uncertainty ranges for ethanol, RME and F-T diesel processing (see Table B1).

References

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

Crutzen, P.C., Mosier, A.R., Smith, K.A., Winiwarter, W., 2007. N_2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. Atmospheric Chemistry and Physics Discussions 7, 11191–11205.

EC, 2000. Commission of the European Communities 2000. Green Paper—Towards a European Strategy for the Security of Energy Supply. COM(2000) 769 final. Brussels 29.11.2000.

EC, 2003. Directive 2003/30/EC of the European Parliament and of the Council of May 2003 on the promotion of the use of biofuels or other renewable fuels for transport. Brussels 17.5.2003.

EC, 2006. Commission of the European Communities 2006. Communication from the Commission—An EU Strategy for Biofuels. COM(2006) 34 final. Brussels 8.2.2006.

EC, 2007a. Commission of the European Communities 2007. Communication from the Commission to the European Council and the European Parliament—An Energy Policy for Europe. COM(2007) 1 final. Brussels 10.1.2007.

EC, 2007b. Commission of the European Communities 2007. Communication from the Commission to the Council and the European Parliament—Renewable Energy Road Map; Renewable energies in the 21st century: building a more sustainable future. COM(2006) 848 final. Brussels 10.1.2007.

EC, 2007c. Commission of the European Communities 2007. Communication from the Commission to the Council and the European Parliament—Biofuels Progress Report. Report on the progress made in the use of biofuels and other renewable fuels in the Member States of the European Union. COM(2006) 845 final. Brussels 10.1.2007.

EC, 2008a. Proposal for a Directive of the European Parliament and of the Council amending Directive 2003/87/EC so as to improve and extend the greenhouse gas emission allowance trading system of the Community. Commission of the European Communities, COM(2008) 16 final, Brussels 23.1.2008.

EC, 2008b. Proposal for a Decision of the European Parliament and of Council on the effort of Member States to reduce their greenhouse gas emissions to meet the Community's greenhouse gas emission reduction commitments up to 2020. Commission of the European Communities, COM(2008) 17 final, Brussels 23.1.2008.

EC, 2008c. Proposal for a Directive of the European Parliament and of the Council on the promotion of the use of energy from renewable sources. Commission of the European Communities, COM(2008) 19 final, Brussels 23.1.2008.

EC, 2008d. Communication from the Commission to the Council and the European Parliament on a First Assessment of National Energy Efficiency Action Plans as Required by Directive 2006/32/EC on Energy End-Use Efficiency and Energy Services—Moving Forward Together on Energy Efficiency. Commission of the European Communities, COM(2008) 11 final, Brussels 23.1.2008.

Edwards, R., Griesemann, J.-C., Larivé, J.-F., Mahieu, V., 2003. Well-to-Wheels Analysis of Future Automotive Fuels and Powertrains in the European Context. Jointly carried out by EUCAR, CONCAWE and JRC/IEA. Well-to-Tank Report Version 1, December 2003.

Edwards, R., Szekeres, S., Neuwahl, F., Mahieu, V., de Santi, G. (Eds.), 2008. Biofuels in the European Context: Facts and Uncertainties. European Commission, Joint Research Centre, European Communities, 2008. 〈http://www.jrc.ec.europa.eu/〉.

Electrowatt-Ekono, 2005. Supply, demand and delivery reliability of wood fuels under the EU Emission Trading System. Study report 60K04773.01-Q060-031. ElectroWatt-Ekono, 2005 (in Finnish).

Elsayed, M.A., Matthews, R., Mortimer, N.D., 2003. Carbon and energy balances for a range of biofuel options. 21/3 Final Report. Sheffield Hallam University, Resources Research Unit, United Kingdom, 341pp.

Fargione, J., Hill, J., Tilman, D., Polasky, S., Hawthorne, P., 2008. Land clearing and the biofuel carbon debt. Science 319 (5867), 1235–1238.

Gregorich, E.G., Rochette, P., VandenBygaart, A.J., Angers, D.A., 2005. Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. Soil & Tillage Research 83, 53–72.

Gustavsson, L., Karlsson, Å., 2002. A system perspective on the heating of detached houses. Energy Policy 30 (7), 553–574.

Holttinen, H., Tuhkanen, S., 2004. The effect of wind power on CO_2 abatement in the Nordic Countries. Energy Policy 32 (14), 1639–1652.

IPCC, 1996a. Climate Change 1995. The Science of Climate Change. Contribution of Working Group I to the Second Assessment Report of the Intergovernmental Panel on Climate Change, Great Britain, 572pp.

IPCC, 1996b. Revised 1996 IPCC guidelines for national greenhouse gas inventories. In: Houghton, J.T., Meira Filho, L.G., Lim, B., Treanton, K., Mamaty, I., Bonduki, Y., Griggs, D.J., Callender, B.A. (Eds.). IPCC/OECD/IEA. UK Meteorological Office, Bracknell. Available: 〈http://www.ipcc-nggip.iges.or.jp/public/gl/invs1.htm〉.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories 〈http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.htm〉.

Kara, M., Syri, S., Lehtilä, A., Helynen, S., Kekkonen, V., Ruska, M., Forsström, J., 2008. The impact of EU CO_2 emissions trading on electricity markets and electricity consumers in Finland. Energy Economics 30 (2), 193–211.

Kirkinen, J., Minkkinen, K., Sievänen, R., Penttilä, T., Alm, J., Saarnio, S., Silvan, N., Laine, J., Savolainen, I., 2007. Greenhouse impact due to different peat fuel utilisation chains—a lifecycle approach. Boreal Environment Research 12, 211–223.

Liski, J., Palosuo, T., Peltoniemi, M., Sievänen, R., 2005. Carbon and decomposition model Yasso for forest soils. Ecological Modelling 189, 168–182.

Mäkinen, T., Soimakallio, S., Paappanen, T., Pahkala, K., Mikkola, H., 2006. Greenhouse gas balances and new business opportunities for biomass-based transportation fuels and agrobiomass in Finland. Technical Research Centre of Finland. VTT Research Notes 2357, Espoo. Finland, 134pp.+app. 19pp. (in Finnish).

Mikhailova, E.A., Bryant, R.B., Vassenev, I.I., Schwaiger, S.J., Post, C.J., 2000. Cultivation effects on soil carbon and nitrogen contents at depth in the Russian Chermozem. Soil Science Society of America Journal 64, 738–745.

Mikkola, H., Pahkala, K., 2008. Energy balance of barley for ethanol production. In: Proceedings of the NJF 405 Seminar, Vilnius, Lithuania, 25–26 September 2007, NJF Report 3 (4), 71–77.

Monni, S., Perälä, P., Regina, K., 2007. Uncertainty in Agricultural CH_4 AND N_2O Emissions from Finland–Possibilities to Increase Accuracy in Emission Estimates. Mitigation and Adaptation Strategies for Global Change 12 (4), 545–571.

Palosuo, T., Wihersaari, M., Liski, J., 2001. Net greenhouse gas emissions due to the energy use of forest residues—impact of soil carbon balance. In: Pelkonen, P., Hakkila, P., Karjalainen, T., Schlamadinger, B. (Eds.), Woody Biomass as an Energy Source—Challenges in Europe. European Forest Institute, Joensuu, Finland, pp. 115–122.

Pingoud, K., Pohjola, J., Valsta, L., Karttunen, K., 2006. Case study: integrated impact of forests and harvested wood products. In: Valsta, L., Ahtikoski, A., Horne, P., Karttunen, K., Kokko, K., Melkas, E., Mononen, J., Pingoud, K., Pohjola, J., Uusivuori, J. (Eds.), Wood in mitigation of climate change. Final report. University of Helsinki, Department of Forest Economics. Reports 39, Helsinki (Chapter 4) (in Finnish). 〈http://honeybee.helsinki.fi/%7Evalsta/carbon/hiililoppuraportti-final.pdf〉.

Reicosky, D.C., Kemper, W.D., Langdale, G.W., Douglas Jr., C.L., Rasmussen, P.E., 1995. Soil organic matter changes resulting from tillage and biomass production. Journal of Soil and Water Conservation 50, 253±61.

Schjønning, P., Munkholm, L.J., Elmholt, S., Jørgen, E., Olesen, J.E., 2007. Organic matter and soil tilth in arable farming: management makes a difference within 5–6 years. Agriculture, Ecosystems & Environment 122, 157–172.

Schlamadinger, B., Edwards, R., Byrne, K. A., Cowie, A., Faaij, A., Green, C., Fijan-Parlov, S., Gustavsson, L., Hatton, T., Heding, N., Kwant, K., Pingoud, K., Ringer, M., Robertson, K., Solberg, B., Soimakallio, S., Woess-Gallasch, S., 2005. Optimizing the greenhouse gas benefits of bioenergy systems. 14th European Biomass Conference, 17–21 October 2005, Paris, France, 4pp.

Searchinger, T., Heimlich, R., Houghton, R.A., Dong, F., Elobeid, A., Fabiosa, J., Tokgoz, S., Hayes, D., Yu, T-H., 2008. Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 319 (5867), 1238–1240.

Statistics Finland, 2005. Energy Statistics 2004. Official Statistics of Finland. Statistics Finland, Helsinki.

Statistics Finland, 2006. Greenhouse gas emissions in Finland 1990–2004. National Inventory Report to the European Union, 15 January 2006. 〈http://tilastokeskus.fi/tup/khkinv/fin_nir_2006.pdf〉.

Statistics Finland, 2008. Greenhouse gas emissions in Finland 1990–2006. National Inventory Report under the UNFCCC and the Kyoto Protocol, 11 April 2008. 〈http://www.stat.fi/tup/khkinv/nir_2008_un_20080418.pdf〉.

VTT, 2006. LIPASTO calculation system for traffic exhaust emissions and energy consumption in Finland. Technical Research Centre of Finland 〈http://lipasto.vtt.fi/indexe.htm〉.

Wihersaari, M., 2005. Aspects on bioenergy as a technical measure to reduce energy related greenhouse gas emissions. VTT Publications 564, VTT, Espoo, 93pp.+app. 71pp. 〈http://virtual.vtt.fi/inf/pdf/publications/2005/P564.pdf〉.

PAPER II

How to ensure greenhouse gas emission reductions by increasing the use of biofuels?

Suitability of the European Union sustainability criteria

In: Biomass & Bioenergy 35, 3504–3513.

Available at www.sciencedirect.com

ScienceDirect

http://www.elsevier.com/locate/biombioe

How to ensure greenhouse gas emission reductions by increasing the use of biofuels? – Suitability of the European Union sustainability criteria

*Sampo Soimakallio**, *Kati Koponen*

VTT Technical Research Centre of Finland, Department of Energy Systems, Tekniikantie 2, P.O. BOX 1000, 02044 VTT, Finland

ARTICLE INFO

Article history:
Received 11 March 2010
Received in revised form
14 April 2011
Accepted 22 April 2011
Available online 23 June 2011

Keywords:
Biofuel
Greenhouse gas emission
Sustainability
Criteria
Life-cycle assessment

ABSTRACT

Biofuels are promoted in many parts of the world. However, concern of environmental and social problems have grown due to increased production of biofuels. Therefore, many initiatives for sustainability criteria have been announced. As a part of the European Union (EU) renewable energy promotion directive (RED), the EU has introduced greenhouse gas (GHG) emission-saving requirements for biofuels along with the first-ever mandate methodology to calculate the GHG emission reduction. As explored in this paper, the RED methodology, based on life-cycle assessment (LCA) approach, excludes many critical issues. These include indirect impacts due to competition for land, biomass and other auxiliary inputs. Also, timing issues, allocation problems, and uncertainty of individual parameters are not yet considered adequately. Moreover, the default values provided in the RED for the GHG balances of biofuels may significantly underestimate their actual impacts. We conclude that the RED methodology cannot ensure the intended GHG emission reductions of biofuels. Instead, a more comprehensive approach is required along with additional data and indicators. Even if it may be very difficult to verify the GHG emission reductions of biofuels in practice, it is necessary to consider the uncertainties more closely, in order to mitigate climate change effectively.

1. Introduction

Transportation accounts for approximately 15% of the global greenhouse gas (GHG) emissions; more than 70% of these come from road transport [1]. The use of gasoline and diesel for road transport will double in the next 25 years and GHG emissions will increase commensurably unless preventative actions are taken [2]. In the past few years several countries, including the United States, China, and the European Union, have announced ambitious policies for promoting the production and use of transportation biofuels (later biofuels) [2]. This is commonly justified by the reduction of GHG emissions through replacing fossil fuels by biofuels. Improvements in energy security, energy independency, and regional employment are other reasons for the particular policies. The supply of biofuels has increased rapidly, though it accounted for only 3% of the total road transport fuel consumption in 2009 [2]. The forecasted share equals 8% in 2035 [2].

A number of recent studies have concluded that the increased production of biofuels may cause significant environmental and social problems [3–12]. Firstly, GHG emission reductions achieved by substituting fossil fuels with biofuels are unclear due to the auxiliary material and energy inputs

* *Corresponding author*. Tel.: +358 207226767; fax: +358 207227604.
E-mail address: sampo.soimakallio@vtt.fi (S. Soimakallio).
0961-9534/$ – see front matter
doi:10.1016/j.biombioe.2011.04.041

required, the direct land-use impact and, in particular, due to indirect impacts such as deforestation. Secondly, other environmental impacts such as nutrient losses due to biomass cultivation and harvesting, and loss of biodiversity may also be significant. Thirdly, the production of biofuels, from raw materials also suitable for food production, has been found to increase food prices and thus cause social problems. Currently, the area of land that is used for biofuel crop production is estimated to be around 1% of the total land used for crop cultivation [13]. However, fulfilling the aggressive targets of increasing the use of biofuels set by various countries will significantly increase the contribution of biofuels in global agricultural land use. The forecasted long-term potential for all bioenergy varies significantly between studies, from some 40 EJ/a to over 1000 EJ/a in the most pessimistic and optimistic scenarios, respectively [14]. The sustainable potential for all bioenergy in 2050 may be only 40–80 EJ/a, which corresponds to 10–20% of fossil energy use today [14].

In order to ensure the sustainable production of biomass and biofuels, several initiatives and certification systems on the sustainability criteria of biofuels and biomass production have been proposed by various organisations and institutions [15]. These initiatives differ from one other. For example they depend on the scope of application; their validity and extent; the variety of environmental, social, and economic aspects considered, and on the conditions set for fulfilling the criteria.

In June 2009, the European Union Directive on the promotion of the use of energy from renewable sources (RED) was published [16]. It establishes a mandatory target to increase the use of renewable energy sources in transportation to 10% in 2020 in the EU. In 2008, the share of renewable energy (probably biofuels) in the overall transportation fuel consumption in the EU is 3.5% [17]. For biofuels and other bioliquids to be accounted in the targets and subsidised, the RED introduces environmental sustainability criteria which need to be met. According to these criteria, the GHG emission reductions compared to fossil comparator shall be at least 35% for biofuels and other bioliquids produced before the end of 2016. From the beginning of 2017, the GHG emission reductions should be at least 50% and from the beginning of 2018, the GHG emission saving should be at least 60% for biofuel production installations where production begins after 1 January 2017. As a part of the sustainability criteria, the RED also introduces a first-ever mandate methodology (later the RED methodology) to calculate the GHG emission balance of biofuels and other bioliquids as well as the GHG emission reduction compared to fossil fuels.

According to the knowledge of the authors the RED methodology has not been critically analysed before. In this paper, we analyse and discuss the suitability of the RED methodology to ensure the GHG benefits of transportation biofuels and other bioliquids (later referred to as biofuels) when fulfilling the biofuel promotion target set by the RED. The paper explores the most critical issues, problems and challenges that are encountered when assessing GHG balances of biofuels in general and compares them with the RED methodology. Based on the most crucial observations we provide suggestions on the way forward.

2. Theoretical framework

2.1. Life cycle assessment

Life Cycle Assessment (LCA) is a methodological framework for estimating and assessing the environmental impacts related to the life cycle of a product system (product or service) [18,19]. Setting the appropriate system boundary and selecting the approach for LCA depends on the goal and scope of the particular study. Two main categories of LCA have been defined: attributional (also defined as descriptive, retrospective) and consequential (also defined as change-oriented, prospective) [20–22]. The Attributional LCA (ALCA) has been defined as a method “to describe the environmentally relevant physical flows of a past, current, or potential future product system”. It can be used to describe GHG emissions of each product manufactured or service produced in the economy at a given point of time. In contrast, the Consequential LCA (CLCA) can be defined as a method that aims to describe how environmentally relevant physical flows would have been or would be changed in response to possible decisions that would have been or would be made. The ALCA reflects the system as it is whereas the CLCA attempts to respond to the question: “What if?”.

2.2. The RED methodology

The RED provides default values for GHG emission reductions (%) compared to fossil reference fuels for a range of biofuels. These default values can be used if GHG emissions from land-use changes can be proved to be equal to or less than zero. In addition, the RED also provides disaggregated default values, separately and as aggregate, for cultivation, fuel processing, and transport and distribution for a range of biofuels expressed as g CO_2-eq./$MJ_{fuel.}$. Disaggregated default values for cultivation can only be used if the raw materials are cultivated outside the European Community, are cultivated in the Community areas included in the specific list referred to in the RED, or are waste or residues from other than agriculture, aquaculture, and fisheries. If the above mentioned conditions are not filled, if the default value for the GHG emission saving from a specific production pathway falls below the required minimum level, or if the default value does not exist, biofuel producers are required to use the RED methodology to show that the actual GHG emission reductions resulting from their production process fulfil the set criteria. In addition, the biofuel producer may always use the actual value instead of the default value [16].

The part C of Annex V of the RED defines the relative reduction in greenhouse gas emissions achievable by replacing fossil fuel comparator by certain biofuel as:

$$EMISSION\ SAVING = (E_F - E_B)/E_F \quad (1)$$

where,

EB = total emissions from the biofuel or other bioliquid; and

EF = total emissions from the fossil fuel comparator.

The formula with details for calculating the actual values for the total emissions from the use of biofuel or other bioliquid (EB) is given in the part C of Annex V of the RED [6]. It takes into account the greenhouse gas emissions from the different phases of the biofuel production from cultivation or collection of raw-material to the use of biofuel. Greenhouse gas emissions are expressed in terms of g CO_2-eq./MJ in Eq. (1). Relating to the implementation of the RED into national legislation of the EU Member States, the European Commission issued two Communications, which include practical guidelines on the implementation of the sustainability system and the associated calculation rules [23], and on voluntary certification systems and default values [24]. In addition, a Decision on the calculation of land carbon stocks in the case of land-use changes was issued [25].

According to the RED methodology, the spatial system boundary includes the biofuel product system from raw material cultivation (crops), harvesting (residues), and collection (waste) to the distribution of biofuel [16]. However, GHG emissions from production of machinery, infrastructure, buildings and plants are excluded. The climate impacts are assessed with the Global Warming Potential (GWP) values for 100 years given by the IPCC [26].

3. Analysis and discussion

3.1. *Conservativeness of the GHG emission default values of the RED*

We compared the GHG emission default values of some biofuels provided in the RED with the figures found from the recent literature. Based on 25 recent studies (see Supporting information for details), the GHG balance figures for various biofuel supply chains vary significantly around the default values provided in the RED (Fig. 1). Some very high GHG emission estimates were found from the literature for biodiesel derived from palm oil and soya oil. However, also lower GHG emission estimates compared to the default values of the RED were found. Based on the literature review it was not possible to conclude in general whether the default values of the RED are conservative or optimistic concerning specific biofuel chains. The variation in the results for specific raw materials may be due to differences in spatial system boundary setting, handling of timing issues, allocation procedure, parameter assumptions, or case-specific features. These issues are discussed in more details in the analysis of the RED methodology in the following chapters.

3.2. *Spatial system boundary*

The RED methodology provides a framework to set the spatial system boundary to calculate actual GHG emissions of biofuels (Fig. 2). The RED methodology seems to follow the principles of ALCA as physical flows relevant for GHG emissions of biofuel product systems are under consideration. Within the defined system boundary, the GHG balances of biofuels depend on the GHG intensity of raw materials and other auxiliary inputs required. Indirect impacts through market effects are not taken into account. Next we discuss the major mechanisms that may result in significant indirect impacts that are excluded from the RED methodology and provide some suggestions to improve the methodology.

3.2.1. *Land and raw material requirement*

As regards crop-based biofuels, the RED encourages the use of land which provides raw materials whose GHG emission

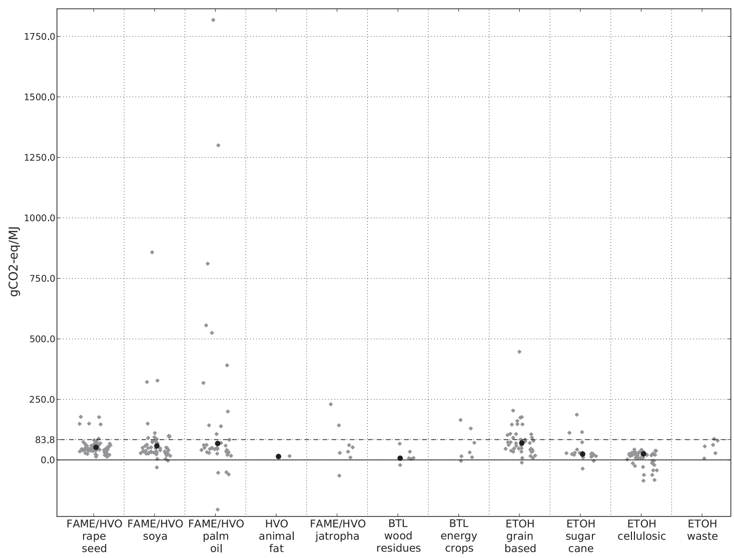


Fig. 1 – GHG balances of different biofuels produced from various raw materials in different regions and using different process technologies (sources presented in the Supporting information). The dotted line illustrates the GHG balance of the fossil reference fuel (gasoline and diesel) including CO_2 emission from fossil fuel combustion in accordance with the RED. The default values of the RED for certain raw materials and technologies are illustrated by black circles. In case the RED provides more than one default value for certain technology route, the maximum value was selected.

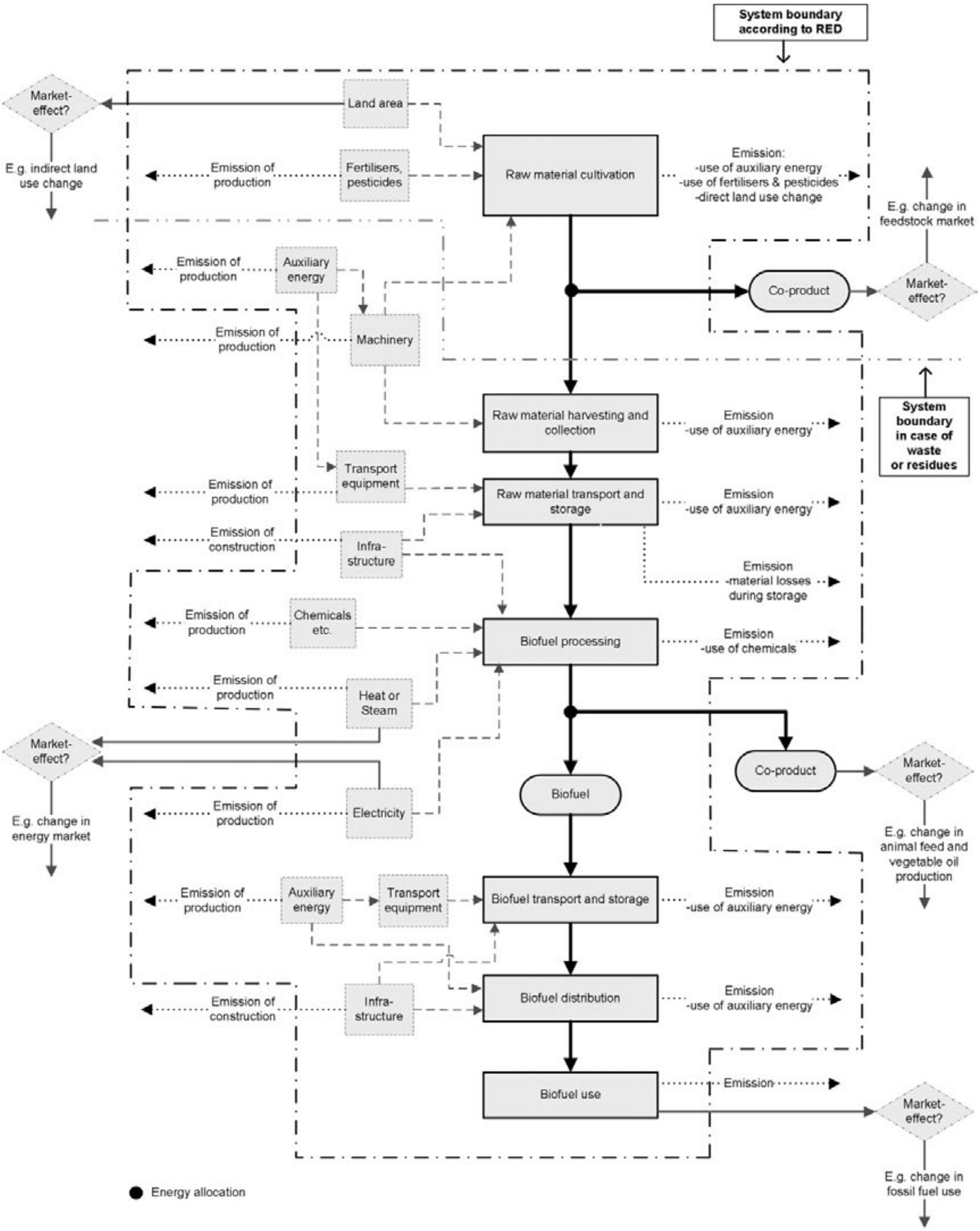


Fig. 2 – Generic illustration of the spatial system boundary according to the RED methodology with possible impacts outside the system boundary.

intensity is low enough to reach the set GHG emission saving requirements. This incentive is reasonable as long as raw materials and land are available for such purposes. Campbell et al. [27] estimated that the global potential for bioenergy on abandoned agriculture lands is less than 8% of the current primary energy demand, which corresponds to some 40% of the current primary energy consumption in personal and commercial transportation. The forecasted population increase together with the expected changes in eating habits will have a significant impact on the requirement of land for food and feed production, and will thus also significantly affect the availability of land for bioenergy production in the future. For example, Hakala et al. [28] estimated that the overall global field bioenergy potential would be roughly 10–30% of the current primary energy use in 2050 depending mainly on the development of the diets. Many recent studies have concluded that the increased production of biofuels from crops or raw materials requiring land that is currently used for agricultural purposes probably increases deforestation and thus results in significant carbon dioxide emissions (e.g [3,10,29]). In practice, these indirect land use changes (iLUC) may be difficult to identify, quantify, and attribute to various economic actions [30]. The European Commission is currently examining how iLUC should be considered [31]. However, at this time it is not clear whether iLUC is included in the RED methodology or handled somehow separately. The EU Member States are deeply divided over the issue of including an iLUC factor in calculation of actual GHG emissions of biofuels. Currently, there is no decision on modifying the RED methodology [31].

In order to reduce the incentives to use raw materials that compete directly with food, the RED encourages, in particular, the production of biofuels from wastes, residues, non-food cellulosic material, and ligno-cellulosic material. The energy contents of the specific biofuels are considered as double when fulfilling the 10% target set for the use of renewable energy in transportation in 2020. The RED methodology encourages the use of residues, such as forest residues, agricultural side streams, and waste streams such as municipal, industrial, or commercial waste as raw materials by considering them free of GHG emissions before collection. However, neither the RED nor the practical guidelines [23] provide a clear definition of 'waste or residues'. There is a possibility that raw materials already utilised might be classified as 'waste or residues' in the context of the RED and thus be promoted for use as biofuels. This may have significant indirect impacts. The definition of 'waste or residue' should be clarified in the context of the RED, and then further analyses of the impacts are possible and certainly required.

Promoting the use of waste or residues in general is reasonable as long as these are generated by other economic activities, when they are not effectively utilised, and when they can be gathered and utilised in a sustainable way. However, avoiding the generation of waste streams is likely to reduce GHG emissions more than generating and utilising waste streams. It is crucial for the incentives provided for the use of waste streams that they do not reduce incentives to avoid generating wastes [32]. Stricter climate policies and ambitious targets to reduce GHG emissions in various economic sectors are likely to increase the competition for waste and residue biomass resources for various end-use purposes; such as power and heat production, material recycling and production, and chemical production. There is scientific evidence that cascading the use of biomass (first as materials and then as energy), when possible, is likely to yield more reductions in GHG emissions than direct energy use [33,34]. Ohlrogge et al. [35], de Santi et al. [10] and Soimakallio et al. [12] concluded that more GHG benefits are gained by using raw materials for power or heat production to substitute coal than by producing more energy intensive liquid biofuels to substitute oil. Increased use for biofuels may thus decrease the availability of raw materials for other purposes and increase the use of more GHG emission-intensive raw materials.

3.2.2. *Other auxiliary inputs*

In addition to land and raw material resources, the production of biofuels requires other auxiliary inputs, such as energy. Koponen et al. [36] studied the GHG balances of lignocellulosic-based ethanol derived from commercial and industrial waste streams in Finland in accordance with the RED methodology. Soimakallio et al. [12] studied the GHG balances of biodiesel derived from logging residues and reed canary grass. The production was based on gasification and Fischer-Tropsch (FT) synthesis integrated into pulp and paper mills. Both studies concluded that GHG emission reductions are highly dependent on the GHG emission intensity of the auxiliary energy. Consequently, a way must be found to keep the GHG intensity of the auxiliary energy required in biofuel production low enough to fulfil the target set by the RED. This can be done, for instance, by increasing the use of renewable energy sources with low GHG emission intensity as auxiliary energy inputs to the biofuel production chain. However, as in the competition for land and raw material resources, the requirement of other auxiliary inputs for biofuel production decreases the overall availability of the particular resources. For example, purchasing hydro power for biofuel production decreases its availability for other purposes; as a result, it needs to be replaced by some other form of electricity [37].

3.2.3. *Co-product outputs*

Biofuel production may generate various types and amounts of co-products such as animal feed, power, and heat. Putting the co-products to the market likely influences the use of competitive products. This tends to have a decreasing impact on the overall GHG emissions from biofuel production. In some cases GHG emission reductions due to product substitution may be very significant.

3.2.4. *Possible methods to include indirect impacts*

Introduction of mandatory targets to increase the use of biofuels together with narrow LCA-based approach to ensure that GHG emissions of biofuels do not exceed certain level, increase significantly the risk that various indirect impacts through market mechanisms take place. The higher is the requirement for the GHG emission reduction of biofuels the more raw materials and other auxiliary inputs with low GHG emission intensity as well as the more productive land is probably transferred to serve biofuel production. This way the emissions may be outsourced to other product systems. The use of certain types of average national, regional, or global figures instead of case-specific figures in determining GHG emissions of resource consumption would probably decrease the emission outsourcing effect. In order to encourage biofuel producers to improve GHG balances of their process in a reasonable way, exceptions could be made if the actors can prove that the resources required would not have been used otherwise. However, this may be very difficult in practice.

The consequences of increasing the use of biofuels may be very far-reaching in space and time. Therefore, they are very difficult to be captured exactly, as there is not enough data and sufficient understanding of the phenomenon available. Scenario analyses can be carried out by using various types of modelling such as economic equilibrium models (e.g [38–40],), partial equilibrium energy system models [41,42] and land use models [43]. However, scenarios are always subjective and uncertain due to the significant number of assumptions required. Applying both the simple and more understandable but narrow LCA, and the more complex and comprehensive but more assumption-sensitive scenario analysis might provide more perspectives in selecting biofuels to be promoted.

3.3. *Timing issues of emissions, sinks, and avoided emissions*

The mitigation of climate change requires rapid and effective actions. According to the IPCC, global GHG emissions should be reduced by at least 50–85% by 2050 from their levels in 2000 in order to limit the global mean temperature increase under 2 °C compared to the pre-industrial level [44]. In the RED methodology and many LCA studies (e.g [45].), the GHG emission impacts are considered by a static method. This means that all GHG emissions and sinks are assumed to take place at the same time and they are then equalised over the lifecycle studied. However, exclusion of dynamics of GHG emissions, sinks and avoided GHG emissions is problematic, particularly when they differ significantly over time, which may be the case for many bioenergy options [46]. For annual crops, the carbon that is released during the combustion is accumulated back into the growing biomass and is not an issue. However, if significant pulse GHG emissions for example from deforestation, the destruction of peat swamps, or from other carbon stock losses take place directly or indirectly due to biofuel production, the consideration of the dynamics of GHG emissions and sinks becomes more crucial in terms of reflecting the actual climate impacts. The sequestration of carbon into forests and soils, and the conservation of large carbon pools (such as peat swamps and pristine forests) are comparable options in climate change mitigation with the use of biomass to substitute fossil fuels.

The RED methodology defines a 20-year horizon for considering the impact of GHG emissions from direct land-use changes (dLUC) to be equalised over the period [16]. The direct deforestation of pristine forests and the use of wetlands for the cultivation of biofuel raw materials are not permitted in the RED, which is likely to prevent the use of land with the most significant soil-based carbon dioxide emissions. However, significant dLUC may take place for example due to the clearing of managed forests. In addition, deforestation and the destruction of peat swamps may take place indirectly, causing significant carbon dioxide emissions. If the iLUC issue is intended to be included in the RED in the future, one problem to be resolved is how to handle the time difference between emissions and avoided emissions. Kendall et al. [46] concluded that when aiming to stabilise the atmospheric GHG concentrations at the ambitious level, the actual climate effects of pulse emissions (for example from land-use changes) are significantly underestimated (70–80%) if annualized for many years (10–50a). Consequently, the suitability of the static LCA method, as introduced in the RED, to assess the climate impacts of bioenergy chains with significant time differences between emissions and sinks or avoided emissions can be questioned. Dynamic indicators (e.g [47].) or derivates from dynamic approaches (e.g [48].) would be much more appropriate, at least when there is a considerable time difference between GHG emissions and sinks or avoided emissions.

Changes in forest carbon stocks, including terrestrial and soil carbon stocks, due to forest biomass harvesting are probably the most important issue to be considered concerning GHG emissions from biofuels derived from forest biomass [49]. It is not clear how changes in biogenic carbon stocks related to land management but not land-use change are intended to be considered in the RED methodology. The timing issues and indicators discussed in this chapter are relevant also for these kinds of carbon stock changes.

3.4. *Allocation procedure*

One of the main challenges in the ALCA involves the allocation of the environmental impacts to the different products, since there is not a single objective or superior method to carry out the allocation procedure [20,50]. Various methods have different pros and cons and the choice of the method may have a significant impact on the LCA results [50]. The allocation in the RED methodology is mainly defined to be based on the energy content of the products determined by lower heating value in the case of co-products other than electricity. However, it should be noted that not all the products are used, primarily or in general, for energy production purposes (for example various materials, animal feed). This makes the general suitability of the particular allocation procedure more or less uncertain depending on the end-use purposes of the co-products. In addition, if heat or steam is generated as a co-product of biofuel processing, as is for example the case of FT diesel processing [11], the RED methodology does not define how the allocation should be carried out as heat does not have lower heating value. If no emissions are allowed to be allocated to heat produced and utilised, the methodology does not encourage integration of biofuel production into a system which can utilise the heat (e.g. pulp and paper mill).

When allocation cannot be avoided, and if only one particular allocation method is to be applied, allocation based on economic value of the products could be the most suitable option [51]. Although the method is not necessarily stable due to fluctuations in the price of co-products, it reflects changes in market conditions and thus prevents allocating emissions to co-products that have no economic value or use.

3.5. *Parameter uncertainty*

Besides systematic uncertainty resulting from normative choices (discussed in Sections 3.2–3.4), the GHG balances of biofuels are subject to uncertainties due to a lack of reliable data. Here we refer to this as 'parameter uncertainty'. In any LCA there are always parameters that vary in terms of their certainty and significance on the results. A large range of uncertainty does not necessarily mean major significance in the overall result if the contribution of the parameter on the result is relatively low. However, as regards the GHG balances of biofuels, many of the most uncertain parameters have been assessed to be the most significant as well [12,52].

Nitrous oxide (N_2O) emissions in agriculture constitute a remarkable uncertainty source in the lifecycle GHG balances of many biofuel pathways [10,12,52]. The use of nitrogen fertilizers and the related nitrogen balance and N_2O emissions strongly depend on site-specific aspects such as crop, soil, and climatic conditions. Consequently, it is difficult to identify representative average emission factors.

According to IPCC [53], soil carbon losses may be significant for agricultural biomass cultivation based on ploughing. In addition, soil organic carbon is an important determinant of soil fertility and to a certain degree, crop productivity has a positive effect on the soil organic matter content [54]. Similarly, the harvesting of logging residues and stumps may change the forest carbon and nutrient cycles due to the removal of organic matter and nutrients along with the raw material [55]. Changes in soil carbon balances may vary significantly depending on the soil and biomass type, biomass cultivation and harvesting measures, and on the climatic conditions [56]. In addition to soil carbon losses, erosion and nutrient losses may have a significant impact on land productivity. According to Jason [57], particularly high rates of erosion accompany soy production, especially in areas where long cycles of crop rotation are not implemented. Pengue [58] reported that intensive soybean cultivation has led to massive soil nutrient depletion in Argentina. All these factors cause uncertainty in upcoming yield rates of land and fertilisation requirement that influences the GHG balances of biofuels.

Uncertainty is involved in every parameter required in the calculations of actual GHG emissions of biofuels or relative GHG emission saving compared to fossil fuels. The level of accuracy required in the determination of the parameters and the cut-off criteria to track the upstream emissions of various auxiliary energy and material inputs are open issues. It is unclear how parameter uncertainties are to be considered in the RED methodology. The more careful consideration of the parameter uncertainties in quantifying CO_2 emission reduction in the RED, especially in large-scale projects, is suggested by Chiaramonti & Recchia [59].

3.6. *Emission saving indicator*

In many recent studies, and in the RED methodology, GHG emission reductions resulting from the use of biofuels instead of fossil reference fuels are measured as the difference of the GHG emission balance between the fossil reference fuel and the biofuel compared to the fossil reference fuel (Eq. (1)). Some of the studies take the end use into account [45] while others exclude it. The exclusion of the end use is appropriate if no changes in the emissions compared to the functional unit (for example kilometer driven) can be expected between the fuels compared. This is also the definition of the RED methodology. More importantly, the fundamental problem of this kind of 'relative emission reduction' indicator is the inability to measure the effectiveness of biomass utilisation as a measure to reduce greenhouse gas emissions. Consequently, the GHG emission savings results may look particularly favourable for biofuel processes in which significant amounts of low GHG emission intensive raw materials are used in relation to the amount of biofuel produced. At the same time, another process for converting biomass to biofuel in more energy-efficient way while using more fossil resources may appear unfavourable in terms of the particular indicator. The effectiveness of use of limited resources - biomass and land - is excluded when using this kind of 'relative emission reduction' indicator. Consequently, this particular indicator cannot be used to compare GHG emission reductions between different end use options for biomass, for example transportation biofuel and electricity production. In order to promote the most efficient options of biomass and land use in climate change mitigation, it would be reasonable to measure the GHG emission balances or savings of biofuels in terms of the limiting factors, for example biomass, land area, or money spent [12,60] instead of the 'relative emission reduction' as defined in the RED.

GHG emission for fossil fuel comparator is defined to be the latest available actual average emission for the fossil part of petrol and diesel consumed in the European Community. If no such data is available, the value used shall be 83.8 g CO_2-eq./MJ_{fuel} for transportation biofuels [16]. GHG emission saving from substitution of fossil fuels is also subject to various uncertainties. First, the direct GHG emissions from fossil fuel provision are not naturally exact and may vary due to differences in extraction and transportation of crude oil, fuel processing (mainly flaring), storing, distribution, and dosage of products. However, the variation is typically significantly lower compared to the variation presented for biofuels. For example, Edwards et al. [61] reported approximately $\pm15\%$ variation for fossil gasoline and even lower for fossil diesel provision. Second, fossil fuel energy systems may also result in indirect impacts, such as deforestation due to production of access roads, drilling platforms and pipelines, oil shale and oil sand production, and military security, which may also be significant [30]. However, it is very uncertain how much these indirect impacts can be reduced by replacing a certain relatively minor amount of fossil fuel use by biofuels. Thirdly, it is unclear how much one unit of biofuel actually replaces fossil fuel as such substitution effects are subject to various market mechanisms, and carbon leakage possibilities due to lack of comprehensive, ambitious, and binding GHG emission reduction targets worldwide. This indirect fuel use change issue is explored by Rajagopal et al. [62]. Due to the above mentioned reasons the GHG emission reduction from fossil fuel replacement may vary significantly from the default value given in the RED.

4. Conclusions

The GHG emissions of biofuels calculated in accordance with the RED methodology depend on the case specific features, but more importantly, on the interpretation of the concepts and definitions given in the RED. For example, classification of 'waste or residues' and accuracy in determination of every single parameter required in the calculations are open issues at the moment. Thus, it is not possible to conclude how exactly the GHG emissions will be calculated. However, as discussed in this paper there are serious risks that the RED methodology underestimates the GHG emissions related to biofuel production due to subjective setting of system boundary and other methodological choices.

LCA is a limited approach with various potential sources of uncertainty and variability in input data, scenarios, and models and with no "right" answer. According to Finnveden et al. [20], the uncertainty can be dealt with in the "scientific" way to reduce the uncertainty, the "social" way to discuss the uncertainty in order to find a consensus, and the "statistical" way to incorporate the uncertainty. Lloyd and Ries [63] concluded that qualitative uncertainty analysis in LCA will improve decision making by identifying the likelihood that an alternative will have a lower environmental impact than others or the likelihood of exceeding inventory or impact thresholds. They also concluded that by determining the important contributors to uncertainty in LCA, areas in which an improved understanding is needed will be highlighted. The methods and the data used in LCA should always be subjected to the critical discussion. Therefore, it is reasonable to ask whether the LCA is ready to move from an analysis tool to a decision tool. We conclude that applying the RED methodology to select the biofuels to be promoted in the EU cannot ensure that GHG emissions are reduced.

There is a risk that the RED methodology promotes biofuels with low reduction or even increase in the overall GHG emissions, and prevents biofuels with higher benefits at the same time. It is reasonable to ask whether a biofuel supply chain that exceeds the GHG emission reduction target set in the RED but which is likely to cause significant negative indirect effects through resource competition is a better option to gain incentives than a biofuel supply chain significantly less likely to cause indirect effects but which does not exceed the GHG emission reduction target set in the RED. Consequently, we suggest that the overall calculation methodology for determining GHG emission savings of biofuels should be reconsidered and modified. In addition, the suitability of the principles to select the biofuels to be promoted should be critically assessed.

In order to mitigate climate change, only biofuels resulting in actual GHG emission reductions should be promoted. Certain fundamental principle requirements can be defined for that purpose. Firstly, in order to avoid the significant negative indirect effects, unused raw materials and land area for biomass cultivation should be available. Secondly, the lifecycle GHG impacts of the biofuel product system should be lower than those of fossil reference fuels. Thirdly, biofuel production should not lead to a more ineffective use of fossil fuels. However, ensuring case by case that all the conditions mentioned above are met may be challenging. Thus, the overall use of biomass and land for food, feed, fibre, fuels, and ecosystem services should possibly be based on a comprehensive, integrated, and sustainable action plan. Integrated programs for land use and territorial planning, sectoral policies as well as targeted policy instruments, such as protected area networks were claimed by European Environment Agency (EEA) to tackle trade-offs between many interests for land use in Europe [64]. Also the understanding of the critical issues needs to be improved in order to reduce the most significant uncertainties involved. We finally suggest that in climate change mitigation, more attention should be paid to uncertainties related to various emission reduction measures, in order to promote primarily the most certain ones.

Acknowledgements

The authors wish to acknowledge BIOVAIKU and SUBICHOE projects of the BioRefine programme of the Finnish Funding Agency for Technology and Innovation (Tekes), FOBIT project of the Academy of Finland (124421), the EU Bioenergy Network of Excellence project, and the VTT Technical Research Centre of Finland for financing.

Appendix. Supplementary material

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.biombioe.2011.04.041.

REFERENCES

[1] Baumert KA, Herzog T, Pershing J. Navigating the numbers: greenhouse gas data and international climate policy. World Resources Institute.; 2005.
[2] International Energy Agency. World energy outlook 2010; 2010.
[3] Searchinger T, Heimlich R, Houghton RA, Dong F, Elobeid A, Fabiosa J, et al. Use of U.S. Croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 2008;319:1238–40.
[4] Fargione J, Hill J, Tilman D, Polasky S, Hawthorne P. Land clearing and the biofuel carbon debt. Science 2008;319:1235–8.
[5] Runge CF, Senauer B. How biofuels could starve the poor. Foreign Aff 2007;86(3):41–53.
[6] Mitchell DO. A note on rising food prices. Policy Research Working paper 4682, Development Prospects Group. Washington, DC: World Bank; 2008.
[7] Reijnders L, Huijbregts MAJ. Palm oil and the emission of carbon-based greenhouse gases. J Clean Prod 2008;16:477–82.
[8] Righelato R, Spraclen DV. Carbon mitigation by biofuels or by saving and restoring forests. Science 2007;317:902.
[9] Doornbosch R, Steenblik R. Biofuels: is the cure worse than the disease. Round table on sustainable development. Paris: OECD; 2007.
[10] de Santi G, Edwards R, Szekeres S, Neuwahl F, Mahieu V, editors. Biofuels in the European context: facts and uncertainties. European Commission Joint Research Centre, JRC; 2008.
[11] VTT Research notes 2482. Espoo. In: Soimakallio S, Antikainen R, Thun R, editors. Assessing the sustainability of liquid biofuels from evolving technologies – A Finnish approach; 2009.
[12] Soimakallio S, Mäkinen T, Ekholm T, Pahkala K, Mikkola H, Paappanen T. Greenhouse gas balances of transportation biofuels, electricity and heat generation in Finland—dealing with the uncertainties. Energy Pol 2009;37:80–90.
[13] De Fraiture C, Giordano M, Liao Y. Biofuels and implications for agricultural water use: blue impacts of green energy. Water Pol 2008;10(1):67–81.

[14] Bringezu S, Schütz H, O'Brien M, Kauppi L, Howarth RW, McNeely J. Towards sustainable production and use of resources: assessing biofuels. United Nations Environment Programme (UNEP) & International Panel for Sustainable Resource Management; 2009.
[15] International Biofuels Project Rapid Assessment 22–25 September 2008 Gummersbach, Germany. In: Howarth RW, Bringezu S, editors. Biofuels: environmental consequences and Interactions with changing land use Proceedings of the scientific Committee on problems of the Environment (SCOPE); 2009.
[16] Directive of the European Parliament and of the Council on the promotion of the use of energy from renewable sources. 2009/28/EC. The Official Journal of the European Union, http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2009:140:0016:0062:EN:PDF 5.6.2009.
[17] Eurostat. Share of biofuels in fuel consumption of transport –[tsdcc340], http://epp.eurostat.ec.europa.eu/tgm/table.do?tab=table&init=1&plugin=1&language=en&pcode=tsdcc340; 9 December 2009.
[18] ISO 14040:2006. Environmental management – Life cycle assessment – Principles and framework. International Organization for Standardization (ISO); 2006. 20 p.
[19] ISO 14044. Environmental management – Life cycle assessment – Requirements and guidelines. International Organization for Standardization (ISO); 2006. 46 p.2006.
[20] Finnveden G, Hauschild MZ, Ekvall T, Guinée J, Heijungs R, Hellweg S, et al. Recent developments in life cycle assessment. J Environ Manage; 2009:1–21.
[21] Ekvall T, Tillman AM, Molander S. Normative ethics and methodology for life cycle assessment. J Clean Prod 2005;13: 1225–34.
[22] Curran MA, Mann M, Norris G. The international workshop on electricity data for life cycle inventories. J Clean Prod 2005; 13(8):853–62.
[23] Communication from the Commission on the practical implementation of the EU biofuels and bioliquids sustainability scheme and on counting rules for biofuels. Official Journal of the European Union. 2010/C 160/02.
[24] Communication from the Commission on voluntary schemes and default values in the EU biofuels and bioliquids sustainability scheme. Official Journal of the European Union. 2010/C 160/01.
[25] Commission decision of 10 June 2010 on guidelines for the calculation of land carbon stocks for the purpose of Annex V to Directive 2009/28/EC. Official Journal of the European Union. 2010/335/EU.
[26] IPCC. Climate change. In: Houghton J, Ding Y, Griggs D, Noguer M, Van der Linden P, Xiaosu D, editors. The scientific Basis. Contribution of Working Group I to the Third assessment Report of the Intergovernmental Panel on climate change. Cambridge: Cambridge University Press, http://www.grida.no/publications/other/ipcc_tar/?src=/CLIMATE/IPCC_TAR/WG1/index.htm; 2001.
[27] Campbell JE, Lobell DB, Genova RC, Field CB. The global potential of bioenergy on abandoned agriculture lands. Environ Sci Technol 2008;42:5791–4.
[28] Hakala K, Kontturi M, Pahkala K. Field biomass as global energy source. Agri Food Sci 2009;18(3–4):347–65.
[29] JRC. Indirect land use change from increased biofuels demand. Comparison of models and results for marginal biofuels production from different feedstocks. http://www.jrc.ec.europa.eu/.
[30] Liska AJ, Perrin RK. Indirect land use emissions in the life cycle of biofuels: regulations vs. science. Biofuels Biorpod Bioref 2009;3(3):318–28.
[31] Report from the Commission on indirect land-use change related to biofuels and bioliquids. 811 final. Brussels, 22.12.2010. COM, http://ec.europa.eu/energy/renewables/biofuels/doc/land-use-change/com_2010_811_report_en.pdf; 2010.
[32] Directive of the European Parliament and of the Council on waste. 2006/12/EC. The Official Journal of the European Union, http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2006:114:0009:0021:EN:PDF 27.4.2006.
[33] Pingoud K, Pohjola J, Valsta L. Assessing the integrated climatic impacts of forestry and wood products. Silva Fenn 2010;44(1):155–75.
[34] Valsta L, Lippke B, Perez-Garcia J, Pingoud K, Pohjola J, Solberg B. Use of forests and wood products to mitigate climate change. In: Bravo F, LeMay V, Jandl R, von Gadow K, editors. Managing forest ecosystems: the challenge of climate change, Vol. 17; 2008. p. 137–49.
[35] Ohlrogge J, Allen D, Berguson B, DellaPenna D, Shachar-Hill Y, Stymne S. Driving on biomass. Science 2009;324(5930): 1019–20.
[36] Koponen K, Soimakallio S, Sipilä E. Testing the European Union sustainability criteria for biofuels - case study of waste-derived ethanol. 18th European Biomass Conference and Exhibition, 3–7 May 2010, Lyon, France, 2019–24.
[37] Pereira, MVF. Optimal Scheduling of Hydrothermal Systems – An Overview. Planning and Operation of Electric Energy Systems Proceedings of the IFAC Symposium, Rio de Janeiro, Brazil, 1985, Pergamos Press, Oxford, England; 1986.
[38] Manne A, Mendelsohn R, Richels R MERGE. A model for evaluating regional and global effects of GHG reduction policies. Energy Pol 1995;23(1):17–34.
[39] Nordhaus WD. Roll the DICE Again: the economics of global warming. Yale University; January 28, 1999. Version rice98 pap 121898.wpd.
[40] Nijkamp P, Wang S, Kremers H. Modeling the impacts of international climate change policies in a CGE context: the use of the GTAP-E model. Econ Model 2005;22:955–74.
[41] Klaassen G, Riahi K. Internalizing externalities of electricity generation: an analysis with MESSAGE-MACRO. Energy Pol 2006;35:815–27.
[42] Ekholm T, Soimakallio S, Syri S, Savolainen I, Höhne N, Moltmann S. Effort sharing in ambitious global mitigation scenarios. Energy Pol 2010;38(4):1797–810.
[43] Adams DM, Alig RJ, Callaway JM, McCarl BA, Winnett SM. The forest and agricultural sector optimization model (FASOM): model structure and policy applications. Res. Pap. PNW-RP-495. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station; 1996. 60 pp.
[44] IPCC. In: Metz B, Davidson O, Bosch P, Dave R, Meyer L, editors. Climate Change 2007: Mitigation. contribution of working group III to the Fourth assessment report of the intergovernmental panel of climate change. Summary for policymakers and technical summary; 2007. p. 108. Cambridge.
[45] Edwards R, Larivé J, Mahieu V, Rouveirolles P. Well-to-wheels analysis of future automotive fuels and powertrains in the European context. CONCAWE – EUCAR – JRC -report. Version 2c; March 2007.
[46] Kendall A, Chang B, Sharpe B. Accounting for time-dependent effects in biofuel life cycle greenhouse gas emissions calculations. Environ Sci Technol 2009;43: 7142–7.
[47] Kirkinen J, Palosuo T, Holmgren K, Savolainen I. Greenhouse impact due to the use of Combustible fuels: life cycle Viewpoint and relative Radiative Forcing Commitment. Environ Manage 2008;42:458–69.
[48] Cherubini F, Peters G, Bernsten T, Strømman A, Hertwich E. CO_2 emissions from biomass combustion for bioenergy: atmospheric decay and contribution to global warming. GCB Bioenergy; 2011.

[49] Repo A, Tuomi M, Liski J. Indirect carbon dioxide emissions from producing bioenergy from forest harvest residues. GCB Bioenergy 2011;3(2):107–15.
[50] Ekvall T, Finnveden G. Allocation in ISO 14041 – a critical review. J Clean Prod 2001;9:197–208.
[51] Guinée JB, Heijungs R, Huppes G. Economic allocation: examples and derived decision Tree. Int J LCA 2004;9(1):23–33.
[52] Crutzen PJ, Mosier AR, Smith KA, Winiwarter W. N_2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. Atmos Chem Phys 2008;8:389–95.
[53] IPCC. 2006 IPCC guidelines for national greenhouse gas inventories, http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html; 2006.
[54] Reijnders L. Ethanol production from crop residues and soil organic carbon. Resour Conserv Recycling 2008;52:653–8.
[55] Palosuo T, Peltoniemi M, Mikhailov A, Komarov A, Faubert P, Thürig E, et al. Projecting effects of intensified biomass extraction with alternative modelling approaches. Forest Ecol Manage 2008;255(5–6):1423–33.
[56] Schils R, Kuikman P, Liski J, van Oijen M, Smith P, Webb J, et al. Review of existing information on the interrelations between soil and climate change. ClimSoil, Final Report 16; December 2008.
[57] Jason WC. World Agriculture and the Environment: a commodity-by-commodity guide to impacts and practices. Island Press; 2004. 570 pp.
[58] Pengue W. Transgenic crops in Argentina: the ecological and social debt. Bull Sci Technol Soc 2005;25:314–22.
[59] Chiaramonti D, Recchia L. Is life cycle assessment (LCA) a suitable method for quantitative CO_2 saving estimations? the impact of field input on the LCA results for a pure vegetable oil chain. Biomass Bioenergy 2010; 34(5):787–97.
[60] B. Schlamadinger, R. Edwards, K.A. Byrne, A. Cowie, A. Faaij, C. Green et al. Optimizing the greenhouse gas benefits of bioenergy systems. 14th European Biomass Conference, 17–21 October 2005, Paris, France, 4p.
[61] Edwards R, Larivé J, Mahieu V, Rouveirolles P. Well-to-wheels analysis of future automotive fuels and powertrains in the European context. CONCAWE – EUCAR – JRC -report. Version 3.0, http://ies.jrc.ec.europa.eu/uploads/media/WTT%2520App%202%20v30%20181108.pdf; November 2008.
[62] Rajagopal D, Hochman G, Zilberman D. Indirect fuel use change (IFUC) and the lifecycle environmental impact of biofuel policies. Energy Pol 2011;39(1):228–33.
[63] Lloyd SM, Ries R. Characterizing, propagating, and analyzing uncertainty in life-cycle assessment. A Survey of quantitative approaches. J Ind Ecol 2007;11(1): 161–79.
[64] European Environment Agency (EEA). The European Environment. State and outlook 2010. Land use. http://www.eea.europa.eu.

PAPER III

The complexity and challenges of determining GHG (greenhouse gas) emissions from grid electricity consumption and conservation in LCA (life cycle assessment)

A methodological review

In: Energy 36, 6705–6713.

Energy 36 (2011) 6705–6713

Contents lists available at SciVerse ScienceDirect

Energy

journal homepage: www.elsevier.com/locate/energy

Review

The complexity and challenges of determining GHG (greenhouse gas) emissions from grid electricity consumption and conservation in LCA (life cycle assessment) – A methodological review

Sampo Soimakallio [a,*], Juha Kiviluoma [a], Laura Saikku [b]

[a] VTT Technical Research Centre of Finland, P.O. BOX 1000, FIN-02044 VTT, Finland
[b] University of Helsinki, P.O. BOX 65, FIN-00014 University of Helsinki, Finland

ARTICLE INFO

Article history:
Received 5 April 2011
Received in revised form
15 October 2011
Accepted 19 October 2011
Available online 16 November 2011

Classification:
13.010: Electricity Markets
13.060: Electricity Savings
19.020: Life Cycle Assessment
19.060: CO2-emissions
21.100: Energy and Climate Change

Keywords:
Greenhouse gas emission
Life cycle assessment
Electricity market
Uncertainty

ABSTRACT

The way in which GHG (greenhouse gas) emissions associated with grid electricity consumption is handled in different LCA (life cycle assessment) studies, varies significantly. Apart from differences in actual research questions, methodological choices and data set selection have a significant impact on the outcomes. These inconsistencies result in difficulties to compare the findings of various LCA studies. This review paper explores the issue from a methodological point of view. The perspectives of ALCA (attributional life cycle assessment) and CLCA (consequential life cycle assessment) are reflected. Finally, the paper summarizes the key issues and provides suggestions on the way forward. The major challenge related to both of the LCA categories is to determine the GHG emissions of the power production technologies under consideration. Furthermore, the specific challenge in ALCA is to determine the appropriate electricity production mix, and in CLCA, to identify the marginal technologies affected and related consequences. Significant uncertainties are involved, particularly in future-related LCAs, and these should not be ignored. Harmonization of the methods and data sets for various purposes is suggested, acknowledging that selections might be subjective.

1. Introduction

The production and distribution of electricity consumed makes a significant contribution to the overall GHG balances of various products and processes. Electricity differs significantly from many other energy carriers, as it cannot be stored as such, and is therefore consumed virtually at the same time as it is produced. Electricity can be transmitted for long distances via overhead lines and power cables. Within an electrical network, the consumption and thus also the production typically varies between times of day, seasons, and between years. Furthermore, the electricity production mix varies from one moment to another, and can be very different in different electrical grids. Transmission of electricity between utilities in neighboring regions has been common for many years. Transmission is economically efficient, as it reduces the overall requirement for reserve margins and balances the fluctuations in load within the market area [41]. These specific properties make the assessment of GHG emissions associated with the individual process of consuming or conserving grid electricity a complex and challenging procedure. However, the particular information is highly relevant and required for almost any environmental impact assessment in one form or another.

GHG emissions associated with electricity consumption have been considered in an extensive number of studies related to the cost externalities of electricity consumption [e.g 1–6]., energy system analysis [e.g 6–10]., life cycle assessment (LCA) [e.g 9,11–13], and in the context of CDM (Clean Development Mechanism) project methodologies [14]. The perspectives on environmental impact assessment vary from product or process level to macro level, such as the overall energy system of a country. The appropriate methodology for analyzing the question at hand should be selected accordingly.

LCA is a methodological framework for estimating and assessing the environmental impacts related to the life cycle of a product or process [15,16]. Typically, a LCA study covers the life cycle of a product or process from 'cradle to grave' but may also be limited to

* Corresponding author. Tel.: +358 20 7226767; fax: +358 20 7227604.
E-mail address: sampo.soimakallio@vtt.fi (S. Soimakallio).

0360-5442/$ – see front matter
doi:10.1016/j.energy.2011.10.028

a certain part of the life cycle, for instance the use phase. The LCA is initiated by defining the goal and scope; this is followed by a life cycle inventory, a life cycle impact assessment and an interpretation of the results [17,18]. The ISO standards ISO 14040, 14044 [17,18] guide the basic framework of LCA, but do not provide guidelines on how, in particular, GHG emission estimates of electricity consumption should be determined. The estimates used in LCA of various products may vary significantly, with no clear reasoning behind the assumptions used; which may make the results confusing and disparate. A similar problem has been recognized within cost externality studies [19]. Furthermore, in LCA studies the uncertainty analysis is often lacking or considered only cursorily [20]. The uncertainty in LCA is due to methodological choices, parameters, and models [21]. In addition, variation in the results is due to spatial and temporal variability and variability in objects and sources [21]. The comprehensive uncertainty and sensitivity analysis should consider all the above-mentioned aspects.

The development of LCA has led to a definition of two main LCA categories: *attributional* and *consequential* [16,22]. *Attributional* LCA (hereinafter *ALCA*) has been defined as a method "to describe the environmentally relevant physical flows of a past, current, or potential future product system," [23]. It can be used to calculate the GHG emissions of every product produced in the economy at a given point in time. Thus, the GHG emissions from an appropriate electricity production mix can be attributed to each of the consumption points within the time frame considered, resulting in average emissions for each kWh of electricity consumed.

Consequential LCA (hereinafter *CLCA*) can be defined as a method for describing how environmentally relevant physical flows would have been, or will be, changed in response to possible decisions that would have been or will be made [16,23]. The CLCA methodology often includes the markets affected by decisions [24]. Momentary changes in the consumption of electricity influence the marginal production unit. Therefore, marginal data should be used to describe the impact of such changes. In reality, consequences caused by a decision to change electricity consumption may be far reaching in time and space. These issues may also be taken into account in CLCA [24]. The number of consequential LCA studies has increased recently, but only a few studies have systematically aimed to determine marginal data for electricity consumption [8,9,25]. Marginal emission factors for grid electricity are also applied in CDM projects, dealing with replaced electricity production or electricity efficiency improvements, to earn CER (certified emission reduction) credits under the Kyoto Protocol [26,27].

Curran and colleagues [22] review the issues related to electricity data for life cycle inventories. They summarize the complex matter at a general level, but do not discuss in detail the significance of the selection of the approach, and variation and uncertainty related to the data. Ekvall et al. [23] discuss normative ethics and methodology selection for LCA in general using electricity consumption as a case study. Weber et al. [28] treat the consequences of the results of using various standards, protocols, and reporting guidelines when estimating emission factors for grid-based electricity. Mathiensen et al. [25] and Lund et al. [9] consider the uncertainties related to the identification of the marginal electricity production technology within a market area. Only a few studies have been published overall on the methodological issues and data uncertainties, and a comprehensive picture is still lacking.

This methodological review explores the complexity and challenges of determining GHG emissions from individual processes that consume or conserve grid electricity. The critical issues and uncertainties involved are discussed. The main objective of the paper is to structure the significance related to selecting appropriate methodologies and data sets for various research questions at hand. The viewpoints of ALCA and CLCA approaches are reflected. The examples given are mainly from the EU (European Union), but the same conclusions can be applied to any operating electricity market.

2. Challenges in the assessment procedure

2.1. Determination of GHG emissions for electricity production and distribution

In order to discuss the GHG emissions of electricity consumption, aspects related to various forms of electricity production need to be explored. Direct GHG emissions, namely CO_2, CH_4 and N_2O, are generated in electricity production based on the combustion of fuels like coal, coke, crude-oil-based products, natural gas, peat, wood and other biomass fuels. These emissions are highly dependent on the composition and quality of the fuel (including heating value and moisture content), and the technological characteristics of the power plant (including efficiency). CO_2 emissions are typically the dominant GHG emissions from fuel combustion but CH_4 and N_2O may also be relatively significant for certain technologies [29,30]. According to World Energy Council [31], GHG emissions in terms of CO_2-eq. per kWh electricity produced from fuel combustion are typically in the order of 1000–1300 g for brown coal, 800–1000 g for coal, 600–700 g for heavy fuel oil, and 350–400 g for natural gas condensing power.

Besides direct emissions from fuel combustion, electricity production causes indirect GHG emissions released as part of the supply of the fuels, and production of the infrastructure and power plants. These upstream GHG emissions for electricity production based on fossil fuels may be difficult to assess accurately, but are typically estimated to be much lower (from only a few percent to some 20%) compared to direct emissions from fuel combustion [e.g 31,32]. As fossil fuel combustion dominates electricity production in many countries, the upstream emissions typically constitute a relatively low share of the GHG emissions of the overall electricity production mix of countries [33–35]. For biomass-based electricity, however, the biomass provision and related carbon stock changes in the soil and biomass typically produce the most significant proportion of the associated GHG emissions. According to many recent studies, significant uncertainties are involved in these emissions, and they may even make the GHG performance of biofuels inferior to that of fossil fuels [e.g 36–38].

GHG emissions from electricity production that are not based on fuel combustion, such as wind, hydro, solar and nuclear power, are associated completely with the capital goods and the upstream GHG emissions. According to the World Energy Council [31], GHG emissions in terms of CO_2-eq. per kWh electricity produced are estimated at 7–22 g for wind power, 5–90 for hydro power, 13–104 g for solar power and 3–40 for nuclear power.

When electricity power plant produces multi-products such as power, heat, steam, cooling or refinery products, the problem of emission allocation is encountered. Allocation is a widely recognized and challenging methodological problem in LCA, and the selection of an allocation method typically has a significant impact on the results [39]. This is illustrated in Fig. 1 for a hypothetical coal-fired CHP (combined heat and power production plant). Graus and Worrel [40] studied the impact of various allocation methods on national average GHG emission intensity for a number of countries. They showed that the impact of allocation method is the most significant for Belarus, Denmark, Finland, Kazakhstan, Lithuania, and the Russian Federation. These countries utilize CHP the most in their electricity production. In 2008, electricity produced in CHP plants corresponded to some 10% of the gross electricity production in OECD countries [41], yet the share can be significantly higher for certain individual countries [42].

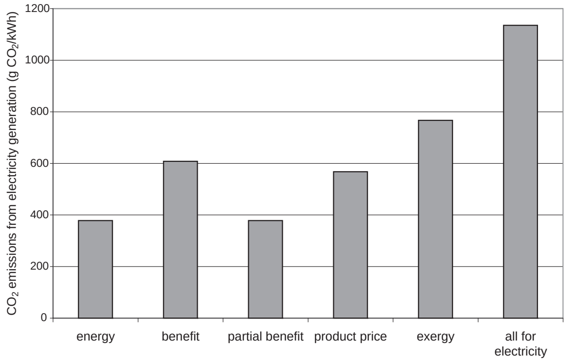


Fig. 1. The influence of various allocation methods on CO_2 emissions from electricity production for a hypothetical coal-fired CHP-plant (power to heat ratio equals 0.5 and the overall efficiency is 90%). The exergy content is assumed to be 1 and 0.24 for electricity and heat, respectively. The economic value of the electricity is assumed to be twice that of heat. In the benefit method, the emissions are allocated to power and heat in relation to the assumed alternative forms of production (condensing power with 39% efficiency and heating boiler with 90% efficiency). In the partial benefit method, emissions are allocated to heat on the basis of the fuel consumption of alternative heat production (90% efficiency), and the remaining share is allocated to power. The absolute numbers are for illustrative purposes only.

GHG emissions resulting from electricity supply - from a power plant to an electricity consumer - also depend on the own electricity use of the power plant and pump storage, heat pumps, and electric boilers, as well as on the transmission, distribution, and transformation losses. Altogether they contributed roughly 14% in OECD countries and 17% world-wide in 2008 [41]. Graus and Worrell [40] found a range of 8–44% within a number of countries. Regarding electricity consumption, these factors should be taken into account. However, it is an open issue how they should be allocated between high, medium, and low voltage consumers, and furthermore, between various transactions. Liberalization of the electricity market allows bilateral contracts between the suppliers and the buyers. In a fully deregulated system, the generators are responsible for their respective loads and their share of transmission losses [43]. This concept leads to confusion in the sharing of transmission loss and the reactive power generated [43]. Gomez et al. [44] concluded that total losses allocated to a transaction may differ significantly depending on the allocation methodology adopted. A common practice involves attributing them to various consumption points based on the average within the region under consideration (e.g. a country). Differentiation between consumers and transactions raises an issue of allocation.

2.2. Determination of the appropriate electricity production mix in ALCA

When estimating the GHG emissions of the electricity consumption of certain process in ALCA, a common practice involves using the average national statistical GHG emissions for electricity production [28]. An example of such a study is Izguierdo et al. [13]. This may be due to the good availability of the annual national statistics, the assumption that the process electricity consumption is constant through time, and the assumption that electricity consumption within a country reflects production within that particular country. Smaller and larger regions than a country are also used [e.g 11]. The BIOGRACE project provides harmonized rules for the calculation of biofuel greenhouse gas emissions in Europe and determines that emissions calculated from grid electricity in Europe should be an average for the EU [45]. The decision to select a smaller or larger region for the determination of the electricity production mix for ALCA is important but arbitrary, as there is no 'correct' choice and there are different types of equity issues involved [28]. Regardless of the choice, the use of annual national or regional statistical average figures in ALCA involves several problems.

The annual national (or regional) average production mix of the electricity may vary significantly from year to year, for instance due to changes in electricity demand, fuel mix, technology portfolio, availability of hydro power, and net imports. For example, in Finland the minimum and maximum annual average CO_2 emissions from electricity production between 1990 and 2002 vary by 20% from the average of the particular period (calculation based on [42]). Consequently, using data for only one statistical year in LCA may significantly reduce the reliability and the applicability of the results to describe the situation for other years.

The variation within a particular year is lost when using annual average figures. The difference in annual and shorter time periods may be highly relevant, in particular when assessing the GHG emissions of a process that operates mainly or only during peak-load hours and when there is significant variation in electricity production mix between peak and base load. For example, Blum et al. [12] studied CO_2 emission savings related to ground source heat pump systems by using an annual average German electricity mix and comparing it with a regional electricity mix for electricity consumption. Similarly, Saner et al. [11] carried out a life cycle assessment of shallow geothermal systems used for heating and cooling by determining the GHG emissions of the electricity consumption by using the annual average electricity mix of Continental Europe and other types of annual average electricity mixes for 2006. Both studies exclude the fact that the electricity consumption of heat pump systems varies significantly between warm and cold seasons. Also, it is very likely that the electricity production mix is different in cold and warm seasons. Thus, examination of the average electricity production mixes studied and the particular consumption curves at a more detailed level, e.g. by months, may probably have influenced the results. When the electricity consumption of a process is not constant throughout a year, it may be reasonable to use figures for shorter time periods instead of annual average figures. However, the availability of the data may generate a practical problem.

Some proportion, minor or major, of the electricity consumed within a country may be produced outside the boarders of the country. Examples of countries where a major proportion of the final electricity consumption is based on imports are Benin, Congo, Lithuania, Luxembourg, Mozambique, the Republic of Moldova, Switzerland, and Togo [46]. Correspondingly, some countries, for example Lithuania, Luxembourg, Mozambique, Paraguay, Slovenia, and Switzerland, export a significant share of their electricity production to other neighboring countries [46]. Therefore, it is justifiable to argue that the average national figures do not reflect the GHG emission profiles of the countries' electricity consumption if they are not adjusted by exports and imports of the electricity. The data is probably available for this kind of adjustment at an annual level [e.g 41]. However, it may prove difficult to find appropriate data which would correspond objectively to the electricity trade by taking into account the precise timing of the trade. The problem caused by the electricity trade between countries can be reduced or avoided by determining a market area larger than a country (e.g. the EU). However, then the electricity consumed within a country does not necessarily reflect the characteristics of the electricity production mix and transmission of that country. As electricity transmission capacity is also limited within a country, it may be reasonable to consider regions smaller than a country in determining the appropriate electricity production mix. Then the problem of considering the electricity transmission between regions is again encountered.

The electricity is typically purchased from the electricity sellers who supply electricity from very different forms of production to different types of customers. Instead of a national or regional average production mix, it may be justifiable to use electricity-seller-specific average figures based on contracts between the seller and the consumer. However, it may be difficult to construct such figures, as the sellers may not be the producers, or they may own only part of certain power plants and the ownership shares may change over time. In the Nordic countries, where one of the world's most sophisticated electricity market exists (Nord Pool), the electricity producers sell a significant amount of the electricity through the exchange to retailers with whom the customers are contracted [47]. This causes a transparency problem between electricity purchased and produced.

In a liberalized electricity market such as that of the EU, consumers can choose their electricity supplier based on prices but also on qualitative criteria such as environmental impacts [48]. 'Green electricity' can be defined as electricity that is produced from renewable sources and that has been differentiated from other electricity products and marketed as being environmentally friendly based on certain criteria [49]. The customers purchasing 'green electricity' might like to consider that the electricity consumed by their processes reflects the mix of the particular 'green electricity' instead of the average mix of the electricity seller or region. This is justified also by the fact that the price of 'green electricity' is typically higher compared to 'regular electricity'. This kind of approach would require the determination of 'contract-based' GHG emission intensities for the electricity. In addition, an allocation problem related to losses is encountered, which has to be resolved.

2.3. *Determining the marginal technology and consequences in CLCA*

ALCA does not reflect GHG impacts of the change in electricity consumption, but it can be used to describe GHG emissions of the average consumption at a given point in time. When the goal is to study the change caused by a particular decision, CLCA can be used. In general, Ekvall and Weidema [24] determined the identification of a marginal technology as a five step process. The current CDM methodology advises project participants to apply six steps in calculating the marginal emission factor to be applied [27]. According to Lund et al. [9] the current 'state-of-the-art' method in CLCA is to identify the long-term change in power plant capacity and to assume that the marginal supply will be fully produced at such a capacity. Traditionally, coal or natural gas has been assumed to reflect the marginal electricity production technology in CLCA (Frees and Weidema [50]; Weidema [51]; Schmidt et al. [52]). The chapters following explore the critical issues related to the determination of appropriate marginal technology and consequences for CLCA.

2.3.1. *Short-term marginal technology*

Instant GHG emissions from electricity production in a market area depend on the existing technology and the relative operational costs of different production units. In an operational electricity market, a marginal increase or decrease in electricity consumption changes the production of the power production unit that is on the margin of the variable cost curve at the time (Fig. 2). If an increase in electricity consumption is greater than the existing marginal power unit can supply, another unit will participate.

The technology serving the short-term (hourly) changes in demand is usually referred as short-term marginal technology. In the CDM methodology this is referred to as "operating margin" [27]. It can vary significantly in time. For example, the marginal production unit may be totally different between day and night and between winter and summer [9]. In the Nordic countries, current marginal production is mainly coal condensing power, but it can also be supplied by other fuels such as gas, oil, peat, waste and wood, and by other technologies such as CHP [47]. Another response to a change in consumption might involve the use of

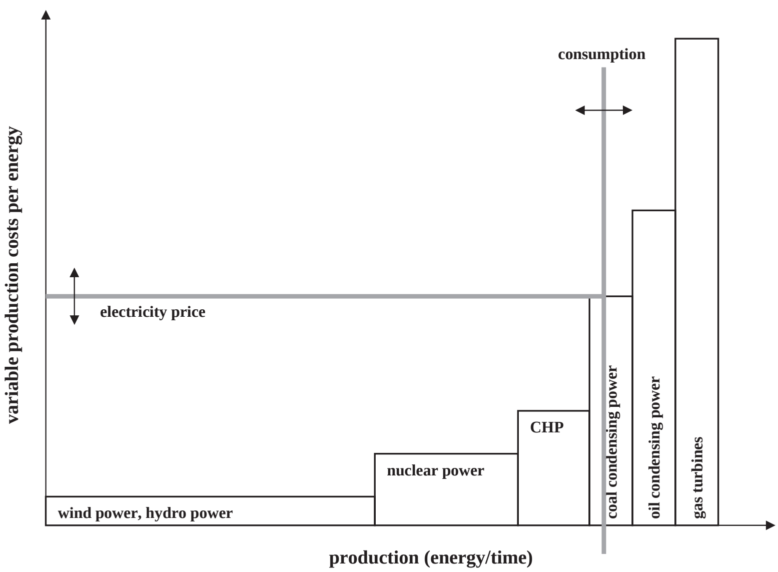


Fig. 2. An illustration of the price formation of electricity in accordance with the order of merit of the power plants that supply instant consumption, which is formed from a number of single consumption points. All of them, regardless of the type of consumption, are connected to the marginal side of production, as a decrease or increase at any consumption point has an impact on the marginal production unit. *Source*: Illustration of production structure is based on Kara et al. [47].

stored energy such as reservoir hydro power. The monetary value of stored water is determined by the forecasted water inflow, the current reservoir level, and the forecasted price of electricity [53]. If the monetary value of the water is higher than the market price of electricity, it is profitable for a hydro power producer to keep storing the water instead of selling it to the grid, and vice-versa. Reservoir hydro power helps the system to use fewer peak power plants and increases the efficiency of the system.

The short-term marginal technology may vary from technologies with nearly zero GHG emissions during operation (e.g. reservoir hydro power) to high GHG emission intensive production forms such as coal condensing power. This should be taken into account when using short-term marginal technology figures in CLCA. For example, it may be necessary to identify short-term marginal technology in order to introduce the electricity saving measurements promptly to cut peak consumption and reduce related GHG emissions efficiently. CDM methodology provides various options to determine the "operating margin" depending on the structure of the electricity generation within the grid and availability of the data [26]. The most accurate determination of "operating margin" within the CDM methodology, being "dispatch data analysis", aims to provide actual data on the short-term marginal technology.

2.3.2. Short-term feedback mechanisms

When comprehensively assessing the actual GHG impacts of changing electricity consumption, the secondary effects caused by the change need to be considered. An increase in electricity consumption may lead to a rise in the price of electricity. The size of the price increase depends mainly on the magnitude of the consumption change, the marginal production affected and potential changes in the production unit in the margin (see Fig. 2.). The price increase may generate additional impacts, such as a reduction in electricity use by some consumers, which can be seen as a negative feedback mechanism [24]. Findings by Alberini et al. [54] suggest that when electricity prices increase, households tend to substitute other inputs for energy and choose less energy-intensive appliances (or homes). On the other hand, Lijesen [55] found a low value for the real-time price elasticity between total peak demand and spot market prices, which may partly be explained by the fact that not all electricity consumers observe the spot market price.

In the short-term, an increase in electricity consumption typically results in a need to use more fuels for electricity production, which may increase fuel prices and furthermore have a reducing impact on overall electricity use. However, Mohammadi [56] found evidence of significant long-run relations only between electricity and coal prices in the U.S. between 1970 and 2007. He also concluded that there is some evidence of unidirectional short-term causality from coal and natural gas prices to electricity prices. The formation of fuel prices is a complicated issue influenced by many socio-economic factors [57]. The feedback mechanism may also be positive, and thus one cannot simply conclude that an increase in fuel consumption unambiguously increases fuel prices.

An increase in electricity consumption also leads to a rise in the absolute CO_2 emissions from electricity production especially when the marginal change is covered by the combustion of fossil fuels. The prevailing climate policy then becomes a limiting factor. For example, in the EU, electricity production is regulated under the EU ETS (EU emission trading scheme) [58]. An increase in CO_2 emissions leads to a rise in the price of emission allowances, as the amount of annual emission allowances available are defined and limited. This may mean that some other actors compensate for the CO_2 emissions resulting from a power plant and satisfy the increased electricity consumption under the EU ETS. Yet, this effect depends on the annual supply and demand of the emission allowances, as well as the mechanisms to invalidate unused emission allowances or transfer them between different years. According to Kara et al. [47], an increase in the emission allowance price also has an incremental influence on the electricity price due to the rise in the production costs of marginal electricity.

It seems obvious that prevailing market conditions and socio-economic issues related to electricity consumption influence electricity production. Eventually, a change in electricity consumption may generate a long chain of positive and negative feedback mechanisms. This makes it difficult to analyze and quantify such impacts. Furthermore, such impacts may be far-reaching, not only in space, but also in time. Thus, a long-term perspective is also required.

2.3.3. Long-term marginal technology

In addition to changes in the current electricity production mix, increased electricity consumption is likely to attract new power plant investments due to increased electricity prices. Investment decisions are further affected by a number of factors reflecting the evolution of the market or by socio-political decisions to regulate emissions. Size and timing of the initial investment together with the subsequent annual cash flows mainly determine the financial performance of a power investment [59]. Changes in electricity consumption can also affect the decisions to retire old power plants from the system. Furthermore, these decisions depend on many other factors, like anticipated fuel prices and other variable costs, as well as investment costs. The simplification of the main interactions of GHG emission impacts from changes in grid electricity consumption is illustrated in Fig. 3.

If 'new consumption' is adequately anticipated before it occurs, there is no unambiguous reason to assign short-term marginal production to this particular consumption. Such a case may occur for example, when 'a new industrial base-load consumption' comes online and a base-load nuclear power plant has been built specifically to anticipate this new consumption. Likewise, an expectation of more air-conditioning in countries with a hot climate is likely to induce investments in peak-load power plants, which will be used during the hours when air-conditioning is needed most. Thus, the expected shape of the consumption profile has implications for the investments required. Adding a constant block of consumption in a traditional electricity system would result in an increase in base-load, intermediate, and peak-load production in the short-term. Yet, in a system where a smart grid has been implemented and consumption is quite flexible, 'the new consumption' could be met with base-load power.

As regards electricity consumption from the grid, the CDM methodology provides two options [26]. First, it advises to calculate the combined margin emission factor of the applicable electricity system by using the procedures in the latest approved version of the "Tool to calculate the emission factor for an electricity system" [27]. This includes the calculation of the weighted average of "operational margin" and "build margin", referring to the group of prospective power plants whose construction and future operation would be affected by the proposed CDM projects [27]. Secondly, default values of 400 and 1300 g CO_2/kWh are provided and can be used under certain strict conditions [26].

Lund et al. [9] showed that marginal change in capacity will have to operate as an integrated part of the total energy system, and therefore, it does not necessarily represent the marginal change in electricity supply, which is likely to involve a mixture of different production technologies. By using detailed ESA (energy system analysis), they assessed that yearly average marginal technologies correspond to a wide range of GHG emission intensity, from 83.3 to 712 g CO_2-eq./kWh, under a business-as-usual 2030 projection of the Danish energy system, depending on the marginal changes in production capacities.

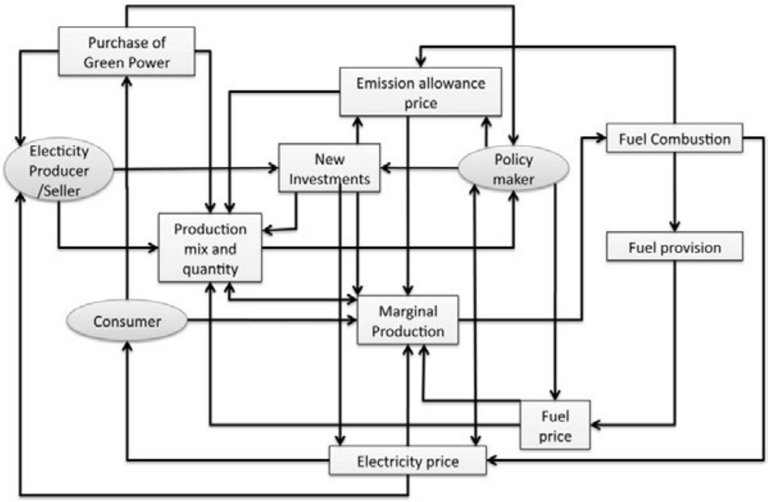


Fig. 3. An illustration of central actors, main factors, and the associated interactions of GHG emission impacts from changes in grid electricity consumption.

Sköldberg and Unger [60] analyzed the energy system and climate effects of adding or reducing 5 TWh of electricity on annual electricity demand in Sweden under various market conditions between 2009 and 2037 using the MARKAL-NORDIC energy system model. According to their results, the impact on new investments generally includes not just a single generation technology but rather a mix of several technologies, and the majority of the effects takes place outside Sweden. They found that the average CO_2 emissions of the change were approximately 700 g CO_2/kWh in several of the scenarios but only 160 g CO_2/kWh if the price of CO_2 was set relatively high (45 EUR/t).

Kiviluoma and Meibom [61] ran a power generation expansion model in order to assess the effect of different flexibility measures on wind power integration costs. In their analysis the cost of wind power was set to result in rather high levels of wind power penetration by 2035. While the time series and existing power plants were from the case of Finland, the results were affected more by the general assumptions about future costs to build and operate different power plants. The annual average CO_2 emissions for the power production varied between 7 and 140 g CO_2/kWh_e in the model runs. They also investigated the change in CO_2 emissions caused by an increase in consumption. They compared a scenario with and without plug-in electric vehicles. The ‘new electricity consumption’ actually reduced the total GHG emissions from electricity production. This was due to the increased flexibility of the power system; a larger fraction of the power production covered by base-load or variable power plants like nuclear and wind power was enabled in the system. In the scenarios where no new nuclear power was built, the total GHG emissions fell on the average by 13 g CO_2/kWh_e with the introduction of the plug-in electric vehicles. If this emission reduction is allocated only to the ‘new consumption’, as might be the case when carrying out CLCA, the decrease is as large as 330 g CO_2/kWh_e. The corresponding figures for the scenarios where nuclear power is added were 1.6 and 41 g CO_2/kWh_e, respectively.

The decisions to curb GHG emissions and increase the share of renewable energy sources are essential in the development of future GHG emissions. If major GHG emission reductions are endorsed and enforced, the electricity sector is likely to bear the greatest share of the burden (e.g. Ekholm et al. [7]). For example in the EU, there are several policy measures affecting the GHG emissions of electricity production directly or indirectly. Such measures include the binding emission reduction directives [58,62], renewable energy directive [63], and targets for energy efficiency improvements [64]. Also, the consumers may purchase ‘green electricity’, aiming to boost the use of renewable energy sources in many EU countries. However, the impact of purchasing ‘green electricity’ on new installations of renewable energy generation capacity can be rather limited in the short-term [48], especially if public policy is strong and feed-in tariffs for renewable energy are widely used [65]. Prevailing and anticipated policy measures significantly affect both the market conditions and the possibilities of consumers to influence the market conditions. Thus, any study attempting to depict future GHG emissions should take policy into account. As long as the development of climate policy is lacking, the long-term marginal technology is also subject to major uncertainties.

3. Conclusions and recommendations

GHG emissions from the production of grid electricity consumed by a certain process are typically assessed in LCA by using statistical average national figures for electricity production mix. However, there are a number of situations where the selection of this particular method is not appropriate. The recent development in LCA has led to the separation of ALCA and CLCA, which have significantly different perspectives and thus also data requirements. Both ALCA and CLCA can be applied to assess GHG emissions from electricity consumption, but only CLCA is appropriate for determining the GHG impacts of a change in consumption. The selection of the approach depends on the goal and scope of the study. The key issues to be considered in an assessment of GHG emissions from electricity consumption are summarized in Table 1.

GHG emissions of a specific power plant depend significantly on the technology and primary energy form used. Furthermore, the system boundaries set for determining individual parameters, consideration of various GHG emission components and choices for other methodological issues, such as allocation, are crucial. Fuel upstream, capital goods, and associated GHG emissions may involve significant uncertainties, and their consideration

Table 1
Key factors and issues when assessing GHG emissions of electricity consumption with attributional (ALCA) and consequential (CLCA) method.

	Attributional LCA (ALCA)	Consequential LCA (CLCA)
Research questions	– How things are (history, current, future perspective)? – Do not reflect change	– What if (history, current, future perspective)? – Reflects change
Short-term technology	– Appropriate average production mix – Spatial dimensions: e.g. power plant, electricity seller, country, market area (including exports and imports) – Temporal dimensions: e.g. instant, seasonal, annual, perennial	– Appropriate marginal production mix – Spatial dimensions: market area with transmission limitations – Temporal dimensions: e.g. instant, seasonal, annual, perennial
Feedback mechanisms	– Not considered	– Market effects (e.g. change in electricity prices and production costs)
Long-term technology	– Estimated future average production mix – The expected development of energy prices, electricity consumption and climate policy significant drivers – Spatial and temporal dimensions as above	– Comparison of GHG emissions with and without the consumption change taking into account power generation investments – The expected development of energy prices, electricity consumption and climate policy significant drivers
Major challenges and suggestions	– Determination of an appropriate production mix and the related GHG emissions – Allocation of emissions between electricity and other products in co-production units – Allocation of losses between consumers and transactions – Harmonization of methodological issues and introduction of data management system to avoid inconsistent GHG emission accounting	– Identification of the marginal technology and related consequences and determination of GHG emissions for relevant power production and other affected activities – Consideration of large uncertainties (e.g. by scenario analysis)

undoubtedly needs to be improved. The determination of GHG emissions of different electricity production forms is the first fundamental challenge encountered in both ALCA and CLCA. Due to various possible sources of uncertainty, it is not possible to objectively determine one single GHG emission figure for any of the power production forms.

Regarding ALCA, one major specific challenge is to define the appropriate production mix of the electricity (Table 1). The key dimensions to be considered are spatial (e.g. national, regional) and temporal (perennial, annual, instant). The selection of the data set may have significant impact on the results. In addition, without a harmonized methodology and data management system, there is a noticeable risk of double-counting either the GHG emissions or the share of certain electricity production forms when considering or comparing the results of various LCA studies. For example, one LCA study may use national figures, whereas another may apply figures of larger or smaller market area. The selection seems to be arbitrary, and it is difficult to determine objectively 'the correct' market area to be considered. Similar problems may be encountered with temporal overlapping, such as between peak-load hours and annual average; and with 'green electricity' if it is not separated from 'the regular electricity' elsewhere.

We conclude that national or regional production mix figures should be adjusted by electricity imports and exports, and they should only be used for analysis concerning electricity consumption at national or regional level, respectively. For history-related ALCA of a single process, figures based on the contract between the electricity seller and the customer with real-time accounting would be the most appropriate production mix figures. A general introduction of this kind of 'contract-based' approach would eliminate the prevailing problem in selecting the market area arbitrarily. However, the use of a harmonized methodology is required in order to deal with the methodological issues encountered (e.g. allocation, system boundaries). In addition, harmonized data management system is needed in order to avoid inconsistencies in the accounting procedure and to maintain the confidentiality of the data. Both of these requirements need further research and general agreements between various stakeholders. Some suggestions on the way forward are already available (e.g. Usva et al. [66]). For future-related ALCA studies, the development of the power production system should be considered by using an appropriate scenario analysis.

Regarding CLCA, the major challenge is to identify the marginal technology, and furthermore, the consequences influenced by the change (Table 1). In its simplistic form, marginal production, affected by the marginal change in the electricity consumption, is identified. Large variations between the affected technologies may occur. We acknowledge the suggestion by Mathiensen et al. [25] of using fundamentally different kinds of affected technologies for this kind of analysis. As the instant marginal GHG emissions of electricity production do not reflect the market effects beyond the immediate change, they are not suitable for describing the related consequences. Such effects may take place in the short term (e.g. increases in electricity price) and long term (e.g. investment decisions). The anticipated development of energy prices, quantity and profile of electricity consumption as well as climate policy are probably the most important market drivers of new investments in electricity production [9].

As changes in the power system are not isolated, electricity consumption and production cannot be separated from each other [9]. When attempting to study the consequences of a decision to change electricity consumption on GHG emissions, an improved understanding of the phenomenon is certainly required. It is important to recognize that not only the electricity production system is affected, but probably many other economic activities as well. Scenarios that depict the changes in economic inputs and outputs can be constructed using economic equilibrium models (e.g. Manne et al., 1995 [67], Nordhaus 1999 [68], Nijkamp et al., 2005 [69]). Yet, due to the complexity of such models, the energy system is typically described in relatively rough terms, limiting the suitability of such models for assessing, for example, GHG emission impacts. Partial equilibrium models for energy systems (e.g. Lund et al. [9], Klaassen & Riahi 2007 [6], Ekholm et al., 2009 [7]) can provide detailed information on the development of energy production to supply external energy demand. By using such models simultaneously, it is possible to create far-flung scenarios to gauge the development of GHG impacts of the economies and various actions. Yet, scenarios always involve a certain degree of uncertainty. Consequently, we suggest that an appropriate number of scenarios are carried out for CLCA in order to provide adequate perspectives on the evolution of the economies, electricity consumption and production as well as GHG emissions under various relevant market conditions.

When it is not possible to carry out a macro-level analysis on the development of future GHG emissions to support LCA, the GHG emissions of various processes or products that consume or conserve electricity need to be assessed individually. Due to the major uncertainties involved, we suggest that a single fixed value for GHG emissions of electricity consumption or conservation should not be used. However, a fixed value may be required in certain cases, such as for determining certified emission reductions of CDM projects related to electricity production or conservation. This can be appreciated but it should be noted that from a scientific viewpoint the use of a fixed value may be highly incorrect. Thus in general, the influence of uncertainties on the overall GHG balance of the concept studied should be analyzed by using an appropriate range of uncertainty. The appropriate range is likely to be lower for ALCA since average values can be used compared to CLCA. Yet, the appropriateness of the range depends on the scope and goal of the study and needs to be carefully considered. If the precautionary principle were to be followed, more conservative rather than optimistic estimates should be used.

Allocation of impacts for various economic activities is always subject to equity issues [23]. CLCA allocates GHG impacts to the decisions considered to cause the impacts. Yet, it is not necessarily fair to separate existing and new electricity consumption when considering the GHG impacts of various decisions. The GHG emissions of new consumption are directly influenced by the existing consumption and vary accordingly. From this point of view, it may be more reasonable to consider that no individual grid electricity consumption can cover the emissions of a particular production. Instead, all consumption should have the same emission intensity based on the average, reflecting the viewpoint of ALCA. On the other hand, a consumer who purchases 'green electricity' should be able to account for the GHG emissions associated with the 'green eletricity' instead of the average emissions, regardless of the actual consequences. Various viewpoints on the equity issues make it impossible to define the most appropriate method over the other ones. After all, selection of the method depends on the goal and scope of the study. This should also be taken into account in the method harmonization processes.

Acknowledgment

The authors wish to acknowledge the GHG Tools project of VTT Technical Research Centre of Finland and the Helsinki University Centre for the Environment (HENVI) for financing, and the anonymous reviewers for their comments.

References

[1] Holdren JP. Integrated assessment for energy-related environmental standards: a summary of issues and findings. Lawrence Berkeley Laboratory; October 1980. LBL-12779.
[2] Hohmeyer O. Social costs of energy consumption: external effects of electricity generation in the Federal Republic of Germany. Berlin: Springer-Verlag; 1988.
[3] European Commission (EC). ExternE: externalities of energy, vol. 1–6. Luxembourg: Office for Official Publications of the European Communities; 1995.
[4] European Commission (EC). ExternE: externalities of energy, vol. 7–10. Luxembourg: Office for Official Publications of the European Communities; 1999.
[5] Oak Ridge National Laboratory (ORNL). Resources for the future (RfF). external costs and benefits of fuel cycles (reports 2–8). Washington: McGraw-Hill Utility Data Institute; 1994–1998.
[6] Klaassen G, Keywan R. Internalizing externalities of electricity generation: an analysis with MESSAGE-MACRO. Energy Policy; 2007:815–27.
[7] Ekholm T, Soimakallio S, Moltmann S, Höhne N, Syri S, Savolainen I. Effort sharing in ambitious global climate change mitigation scenarios. Energy Policy 2009;4:1797–810.
[8] Pehnt M, Oeser M, Swider DJ. Consequential environmental system analysis of expected offshore wind electricity production in Germany. Energy 2008;33: 747–59.
[9] Lund H, Mathiesen BV, Christensen P, Schmidt JH. Energy system analysis of marginal electricity supply in consequential LCA. International Journal of Life Cycle Assessment 2010;15:260–71.
[10] Holttinen H, Tuhkanen S. The effect of wind power on CO_2 abatement in the Nordic Countries. Energy Policy 2004;32:1639–52.
[11] Saner D, Juraske R, Kübert M, Blum P, Hellweg S, Bayer P. Is it only CO_2 that matters? a life cycle perspective on shallow geothermal systems. Renewable Sustainable Energy Review 2010;14:1798–813.
[12] Blum P, Campillo G, Münch W, Kölbel T. CO_2 savings of ground source heat pump systems – a regional analysis. Renewable Energy 2010;35:122–7.
[13] Izquierdo M, Moreno-Rodríguez A, González-Gil A, García-Hernando N. Air conditioning in the region of Madrid, Spain: an approach to electricity consumption, economics and CO_2 emissions. Energy 2011;36:1630–9.
[14] United Nations Framework Convention on Climate Change. Clean development mechanism (CDM), http://cdm.unfccc.int/; 19.8.2011.
[15] Rebitzer G, Ekvall T, Frischknecht R, Hunkeler D, Norris G, Rydberg T, et al. Life cycle assessment Part 1: framework, goal and scope definition, inventory analysis, and applications. Environment International; 2004:701–20.
[16] Finnveden G, Hauschild MZ, Ekvall T, Guinée J, Heijungs R, Hellweg S, et al. Recent developments in life cycle assessment. Journal of Environmental Management 2009;1:1–21.
[17] ISO. ISO 14040. Environmental management – life cycle assessment – principles and framework. International Organization for Standardization (ISO); 2006a. 20 p.
[18] ISO. ISO 14044. Environmental management – life cycle assessment – requirements and guidelines. International Organization for Standardization (ISO); 2006b. 46 p.
[19] Sundqvist T. What causes the disparity of electricity externality estimates? Energy Policy 2004;32:1753–66.
[20] Williams ED, Weber CL, Hawkins TR. Hybrid framework for managing uncertainty in life cycle inventories. Journal of Industrial Ecology 2009;6:928–44.
[21] Huijbregts MAJ. Uncertainty and variability in environmental life-cycle assessment, http://www.ru.nl/contents/pages/13885/binnenwerkhuijbregts1.pdf; 2001.
[22] Curran MA, Mann M, Norris G. The international workshop on electricity data for life cycle inventories. Journal of Cleaner Production 2005;8:853–62.
[23] Ekvall T, Tillman A-M, Molander S. Normative ethics and methodology for life cycle assessment. Journal of Cleaner Production; 2005:1225–34.
[24] Ekvall T, Weidema B. System boundaries and input data in consequential life cycle inventory analysis. The International Journal of Life Cycle Assessment 2004;3:161–71.
[25] Mathiesen BV, Munster M, Fruergaard T. Uncertainties related to the identification of the marginal energy technology in consequential life cycle assessments. Journal of Cleaner Production; 2009:1331–8.
[26] United Nations Framework Convention on Climate Change. Clean development mechanism (CDM). methodological Tool "tool to calculate baseline, project and/or leakage emissions from electricity consumption (Version 01)"; 19.8.2011.
[27] United Nations Framework Convention on Climate Change. Clean development mechanism (CDM). methodological Tool "Tool to calculate the emission factor for an electricity system (Version 02.20)"; 19.8.2011.
[28] Weber C, Jaramillo P, Marriott J, Samaras C. Life cycle assessment and grid electricity: what do we Know and what can we Know? Environmental Science & Technology 2010;6:1895–901.
[29] Tsupari E, Monni S, Tormonen K, Pellikka T, Syri S. Estimation of annual CH_4 and N_2O emissions from fluidised bed combustion: an advanced measurement-based method and its application to Finland. International Journal of Greenhouse Gas Control 2007;3:289–97.
[30] Tsupari E, Monni S, Pipatti R. Non-CO2 greenhouse gas emissions from boilers and industrial processes. Evaluation and update of emission factors for the Finnish national greenhouse gas inventory; 2005. VTT Research Notes 2321. 82 p.+app. 24 p.
[31] World Energy Council. Comparison of energy systems using life cycle assessment. A Special Report of the world energy council; 2004.
[32] Frischknecht R, Althaus HJ, Bauer C, Doka G, Heck T, Jungbluth N, et al. The environmental relevance of capital goods in life cycle assessments of products and services. International Journal of Life Cycle Assessment 2007;11:1–11.
[33] Kim S, Dale BE. Life cycle inventory information of the United states electricity system. International Journal of Life Cycle Assessment 2004;10:294–304.
[34] Santoyo-Castelazo E, Gujpa H, Azapagic A. Life cycle assessment of electricity generation in Mexico. Structural analysis of electricity consumption by productive sectors. The Spanish Case Energy 2011;36:1488–99.
[35] Lee KM, Lee SY, Hur T. Life cycle inventory analysis for electricity in Korea. Energy 2004;29:87–101.
[36] Jungbluth N, Chudacoff M, Dauriat A, Dinkel F, Doka G, Faist Emmenegger M, et al. Life cycle inventories of Bioenergy. Ecoinvent report No. 17, v2.0. Uster, CH: ESU-services. retrieved from: www.ecoinvent.org; 2007.
[37] de Santi G. In: Edwards R, Szekeres S, Neuwahl F, Mahieu V, editors. Biofuels in the European context: facts and uncertainties. European Commission Joint Research Centre, JRC; 2008.
[38] Soimakallio S, Mäkinen T, Ekholm T, Pahkala K, Mikkola H, Paappanen T. Greenhouse gas balances of transportation biofuels, electricity and heat generation in Finland – dealing with the uncertainties. Energy Policy 2009;1:80–90.
[39] Ekvall T, Finnveden G. Allocation in ISO 14041-a critical review. Journal of Cleaner Production; 2001:197–208.

[40] Graus W, Worrel E. Methods for calculating CO2 intensity of power generation and consumption: a global perspective. Energy Policy 2011;39:613–27.
[41] 2010 with 2009 data IEA. Electricity Information. IEA statistics. OECD/IEA; 2010.
[42] Statistics Finland. Energy statistics yearbook 2008. Official Statistics of Finland; 2009.
[43] Bhuiya A, Chowdhury N. Allocation of transmission losses in a deregulated power system network. Proceedings of the 1999 IEEE Canadian Conference on electrical and computer Engineering Shaw Conference Center. Alberta, Canada: Edmonton; May 9-12 1999.
[44] Gomez A, Riquelme JM, Gonzalez T, Ruiz EA. Fair allocation of transmission power losses. Power Systems. IEEE Transactions on 2000;15:184–8.
[45] BIOGRACE. BioGrace calculation rules. Version 1. www.biograce.net.
[46] International Energy Agency (IEA). Energy balances database. Paris: IEA; 2010.
[47] Kara M, Syri S, Lehtilä A, Helynen S, Kekkonen V, Ruska M, et al. The impacts of EU CO_2 emissions trading on electricity markets and electricity consumers in Finland. Energy Economics 2008;2:193–211.
[48] Markard J, Truffer B. The promotional impacts of green power products on renewable energy sources: direct and indirect eco-effects. Energy Policy; 2006:306–21.
[49] Salmela S, Varho V. Consumers in the green electricity market in Finland. Energy Policy; 2006:3669–83.
[50] Frees N, Weidema BP. Life cycle assessment of packaging systems for beer and soft drinks. energy and transport scenarios. Copenhagen: Miljøstyrelsen (Danish EPA); 1998.
[51] Weidema B. Market information in life cycle assessment. Environmental Project No. 863. Copenhagen: Danish Environmental Protection Agency; 2003.
[52] Schmidt AC, Jensen AA, Clausen AU, Kamstrup O, Poslethwaite D. A comparative life cycle assessment of building insulation products made of stone wool, paper wool and flax. Part 1: background, goal and scope, life cycle inventory, impact assessment and interpretation. International Journal of Life Cycle Assessment 2004;9:53–66.
[53] Pereira MVF. Optimal Scheduling of Hydrothermal systems – an Overview. Planning and operation of electric energy systems Proceedings of the IFAC Symposium, Rio de Janeiro, Brazil. Oxford, England: Pergamon Press; 1985. 1986.
[54] Alberini A, Gans W, Velez-Lopez D. Residential consumption of gas and electricity in the U.S.: the role of prices and income. Energy Economics. in press.
[55] Lijesen MG. The real-time price elasticity of electricity. Energy Economics 2006;29:249–58.
[56] Mohammadi H. Electricity prices and fuel costs: long-run relations and short-run dynamics. Energy Economics 2009;31:503–9.
[57] Shafiee S, Topal E. A long-term view of worldwide fossil fuel prices. Applied Energy 2010;87:988–1000.
[58] European Community. Directive 2009/29/EC of the European parliament and of the council of 23 April 2009 amending Directive 2003/87/EC so as to improve and extend the greenhouse gas emission allowance trading scheme of the Community. Official Journal of the European Union; 5.6.2009. 2009.
[59] Laurikka H, Koljonen T. Emissions trading and investment decisions in the power sector – a case study in Finland. Energy Policy 2006;9:1063–74.
[60] Sköldberg H, Unger T. Effekter av förandrad elanvänding / elproduktion –Modellberäkningar. [Impacts of changed electricity use / production – model calculations]. Elforsk rapport 08; 30. April 2008.
[61] Kiviluoma J, Meibom P. Influence of wind power, plug-in electric vehicles, and heat storages on power system investments. Energy; 2010:1244–55.
[62] Decision No 406/2009/EC of the European Parliament and of the Council of 23 April 2009 on the effort of Member States to reduce their greenhouse gas emissions to meet the Community's greenhouse gas emission reduction commitments up to 2020. Official Journal of the European Union 5.6. 2009.
[63] 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. Official Journal of the European Union 5.6. 2009.
[64] Proposal for a directive 2009/28/EC of the European parliament and of the council on energy efficiency and repealing directives 2004/8/EC and 2006/32/EC. Brussels: European Commission, COM; 22.6.2011 (2011) 370 final.
[65] Wüstenhagen R, Bilharz M. Green energy market development in Germany: effective public policy and emerging customer demand. Energy Policy; 2006:1681–96.
[66] Usva K, Hongisto M, Saarinen M, Nissinen A, Katajajuuri J-M, Perrels A, et al. Towards certified carbon footprints of products - a road map for data production. Climate Bonus project report (WP3). VATT Research Reports 143:2. Helsinki: Government Institute for Economic Research; 2009.
[67] Manne A, Mendelsohn R, Richels R. Merge a model for evaluating regional and global effects of GHG reduction policies. Energy Policy 1995;1:17–34.
[68] Nordhaus WD. Roll the DICE again: the economics of global warming. Version rice 98 pap 121898.wpd. Yale University; January 28, 1999.
[69] Nijkamp P, Wang S, Kremers H. Modeling the impacts of international climate change policies in a CGE context: the use of the GTAP-E model. Economic Modelling; 2005:955–74.

PAPER IV

CO_2 emissions attributed to annual average electricity consumption in OECD (the Organisation for Economic Co-operation and Development) countries

In: Energy 38(1), 13–20.

Energy 38 (2012) 13–20

Contents lists available at SciVerse ScienceDirect

Energy

journal homepage: www.elsevier.com/locate/energy

CO_2 emissions attributed to annual average electricity consumption in OECD (the Organisation for Economic Co-operation and Development) countries

Sampo Soimakallio [a,*], Laura Saikku [b]

[a] VTT Technical Research Centre of Finland, P.O. Box 1000, FIN 02044 VTT, Finland
[b] University of Helsinki, Department of Environmental Sciences, P.O. Box 65, FIN 00014 University of Helsinki, Finland

ARTICLE INFO

Article history:
Received 20 July 2011
Received in revised form
27 December 2011
Accepted 29 December 2011
Available online 26 January 2012

Keywords:
Electricity
Trade
CO_2 emission
Life cycle assessment
Uncertainty
Climate policy

ABSTRACT

When regulating GHG emissions at the country or product level, it is critical to determine the GHG emissions from electricity consumption. In this study, we calculated production-based and consumption-based CO_2 emission intensities of electricity for the OECD (the Organisation for Economic Co-operation and Development) countries during 1990–2008. We examined the impact of annual development, allocation procedure in combined heat and power production, and electricity trade on CO_2 emissions. The studied factors significantly, yet highly variably, influenced the results for many countries. The consumption-based CO_2 emission intensity of electricity differed significantly from the production-based intensity for some European OECD countries such as Switzerland, Norway, Slovakia, and Austria. As the use of the production-based method in assessing, verifying, and monitoring the GHG performance of specific products can be highly misleading, the use of consumption-based methods are preferable. The absolute value of CO_2 emissions embodied in electricity net imports accounted for more than 5% of the overall national CO_2 emissions in at least some of the years studied for 13 European countries. The electricity trade and the related GHG emission leakage may increase in the future if effective emission reduction and regulation measures are not more widely implemented.

1. Introduction

Ambitious climate change mitigation requires significant changes in many economic sectors, in particular in the production and consumption of energy [1]. As an energy carrier, electricity plays a fundamental role in modern society and is a mainstay of the worldwide manufacturing industry. While the consumption of primary energy has doubled since the early 1970s, electricity consumption has increased almost fourfold [2,3]. In 2005, CO_2 emissions from fuel combustion in power generation constituted approximately one quarter of all anthropogenic GHG (greenhouse gas) emissions globally [4]. According to many scenarios (e.g.[5]), the electrification of society is set to continue.

Power has been increasingly traded between nations [6,7]. The transfer of electricity between utilities in neighbouring regions has been common practice for many years due to its economic efficiency, which derives from reduced overall requirement for reserve margins and balanced load fluctuations within the market area [6]. In 2008, OECD countries consumed 9244 TWh electricity, imported 372 TWh and exported 360 TWh [6]. Electricity trading between distant locations is limited due to transmission losses. For many countries, however, imported electricity accounts for a significant proportion of total electricity consumption. OECD countries for which imports accounted for more than 30% of total electricity consumption in 2008 were Luxembourg, Switzerland, Denmark, Hungary, Slovenia, Slovakia, and Austria [6]. Furthermore, new transmission system operator investments are underway, with those of European significance corresponding to more than 12% of the existing network until 2020 [8]. In addition, a unified electricity grid for Europe and North Africa by 2050 has been envisioned [9]. Electricity trading is, consequently, likely to increase.

Analysis of the development of GHG emissions of nations and product systems can be carried out by using both prospective (scenario) and retrospective perspectives. In order to understand the feasibility of certain GHG emission development paths, the potential impacts of various technologies and structural changes in the energy system need to be assessed. In this kind of prospective assessment procedure, methods such as consequential life cycle assessment [10,11] and system-level modelling of energy systems [11–13], land use [14], and economies [15], can be used. The assessment of historic GHG emissions by means of retrospective analysis is also important, both to obtain information on trends and for regulation at the country and product level.

* Corresponding author. Tel.: +358 20 7226767; fax: +358 20 7227604.
E-mail address: sampo.soimakallio@vtt.fi (S. Soimakallio).

0360-5442/$ – see front matter © 2012 Elsevier Ltd. All rights reserved.
doi:10.1016/j.energy.2011.12.048

The Kyoto Protocol sets binding GHG emission targets for the period 2008–2012 for those industrialised countries which have ratified the Protocol [16]. The GHG emission reduction targets set under the Kyoto Protocol and the further targets that are currently being negotiated are based on annual production-based GHG emissions within nations [16,17]. The trade of goods, however, may have a significant influence on the development of country-specific GHG emissions associated with the consumption of goods and services [18–20]. Peters et al. [20] estimated that, in 2008, approximately 7.8 Gt of embodied CO_2 emissions were shifted around the globe due to international trade. This is 26% of global CO_2 emissions. Net fossil CO_2 emission transfers from developing to developed countries increased from 0.4 to 1.6 Gt CO_2 during 1990–2008 [20]. GHG emission leakage occurs when the consumption of goods and related production are geographically separated. The risk of significant emission leakage between countries exists at least as long as a comprehensive and effective climate convention is lacking.

One solution to reduce significant emission leakage could be the introduction of consumption-based emission targets for countries or products. This would require the determination of emissions over the life cycle of products. Different types of voluntary standards and criteria-based GHG emission performance rules covering the life cycle of various products have been developed and implemented in recent years [21–24]. In 2009, the EU (European Union) introduced the first ever mandatory criteria for product-based life cycle GHG emissions performance: for transportation biofuels and other bioliquids [25]. Similar binding criteria may also be applied to other products in the future.

Regarding electricity consumption within the monitored product systems, the determination of related GHG emissions is a key issue. The GHG emissions associated with electricity consumption may vary significantly depending on the way the electricity is produced and how the related GHG emissions are determined. Typically, electricity is purchased from the electricity grid, which is formed between a number of power plants and consumption points with various transmission, distribution, and transformation connections. The GHG emissions from final electricity consumption result from fuel combustion and provision; the production and construction of power plants, other capital goods and infrastructure; and from electricity transmission and distribution losses. The relative contribution of these sources varies significantly depending on the form of electricity production [26,27]. However, with respect to GHG emissions from the current electricity production mix, CO_2 emissions from fuel combustion contribute most significantly to the life cycle GHG emissions of electricity production and are the most reliably evaluated [27,28]. Besides the production mix, the choice of allocation method for CHP (combined heat and power production) is essential when assessing the CO_2 emissions of power production from CHP [4]. Furthermore, the consideration of electricity trading is of significant importance, yet it is often ignored.

Emissions embodied in trade have been studied by means of global environmentally extended input–output analyses, including electricity among other goods and services [18–20]. These analyses derive data on trade, economic input–output by sectors, energy consumption, and CO_2 emissions by region and sectors from the GTAP (Global Trade Analysis Project), which compiles primary data from voluntary contributions by each region [29]. Thus, trade is determined based on monetary exchanges. Although the GTAP is widely used in economic analyses, it includes significant uncertainties, such as the currency and quality of primary data, and the unknown magnitude of adjustments made by the GTAP [18,19]. Consequently, such analyses cannot provide accurate data for CO_2 emissions associated with the electricity trade between nations.

Previous studies have examined the GHG emissions of single electricity production technologies [27], the impact of allocation method on CO_2 emissions from CHP (e.g. [4,30]), and the uncertainty of CO_2 emission intensities at various geographic levels in the continental US [31]. Also, the role of international trade on GHG emissions in general has been studied (e.g.[18]). However, according to the knowledge of the authors, the above-mentioned issues have not been studied comprehensively and transparently together in a wider extent for a range of countries. In this paper, we study the role of CO_2 emissions embodied in the electricity trade between nations with respect to annual national CO_2 emissions of electricity consumption. The aim of the paper is to provide information on country-specific electricity emissions for use in a) assessing, verifying, and monitoring the GHG performance of specific products, and b) international climate policy making regarding emission leakage between countries. Data on the production and fuel mix and associated CO_2 emissions, own energy consumption of power plants, distribution and transformation losses, as well as imports and exports of electricity are readily available for various countries. However, detailed data on electricity trading between countries of origin and destination exist only for the OECD countries. Here, we present the estimates for CO_2 emissions from fossil fuel combustion embodied in electricity trade for the 30 OECD countries in 1990, 1995, and 2000–2008. Chile, Estonia, Israel, and Slovenia, which have since been accepted as OECD members in 2010, are not considered in this paper.

2. Material and methods

We examined the CO_2 emission intensity of electricity consumption in the studied countries by both ignoring and considering the CO_2 emissions embodied in the electricity trade; i.e. we estimated the production-based and consumption-based CO_2 emissions of countries. In both cases, the final consumption of electricity was determined by subtracting own use of electricity by power plants, electricity used for heat pumps, electric boilers, and pumped storage, as well as transmission and distribution losses and energy industry consumption of electricity from the net production. First, we calculated the annual production-based CO_2 emission intensity of electricity (g CO_2/kWh$_e$) by determining the total CO_2 emissions from fuel combustion in power production and dividing this by the total amount of electricity produced and transferred to consumption points within a country. In this approach, it was assumed that electricity imports to a country have the same CO_2 emission intensity as the electricity produced within the particular country. Secondly, we calculated the CO_2 emissions embodied in electricity trade and estimated the consumption-based CO_2 emission intensity of electricity (g CO_2/kWh$_e$).

The annual national production-based and consumption-based CO_2 emission intensity of electricity were calculated using equations (1)–(6). In the calculations, we used the latest available data from the IEA (International Energy Agency). The CO_2 emissions from fuel combustion, categorised as electricity output from main electricity producers, autoproducers, and combined heat and power producers, as well as own use of electricity, were taken from the IEA database 'CO_2 emissions from fuel combustion' [3]. The data for electricity production, distribution and transformation losses, imports, exports, and final consumption, as well as electricity and heat production in CHP plants were taken from the IEA database 'Energy Balances' [6]. The data for bilateral electricity trade of the OECD countries were taken from the IEA publication 'Electricity Information' [7], in which electricity is considered to be imported or exported when it has crossed the national territorial boundaries of the country (if electricity is transited through a country, the amount is shown as both an import and an export). The overall national CO_2 emission data were taken from the UNFCCC (United Nations

Framework Convention on Climate Change) [32]. We present the results for eleven years: 1990, 1995, and 2000–2008. National consumption-based CO_2 emission intensities of electricity were calculated only for those OECD countries that trade electricity. Isolated regions do not trade electricity, as they are currently constrained by thermodynamics and the economy of transmission. Thus, for the island nations of Iceland, Japan, Australia, and New Zealand, and also for Korea, production-based emission intensities correspond to consumption-based emission intensities. As some OECD countries import electricity from non-OECD countries, we calculated the production-based CO_2 emission intensity of electricity supply for the non-OECD countries in question using the IEA databases mentioned above. In cases where the origin of electricity import was not known, we applied the production-based CO_2 emission intensity of OECD average. The particular emission intensity was also applied for electricity imports from Luxembourg to Germany between 1990 and 2001 due to lack of reliable data.

The CO_2 emission intensity of electricity production varies throughout the year. Thus, in practice, the emissions embodied in the electricity trade are influenced by the moment of trade. However, data on electricity trade with countries of origin and destination are only available at the annual level. Consequently, we assumed that the production of electricity consumed within and exported from a particular country have the same CO_2 emission intensity. However, quarterly data are available on national electricity production and related combustible fuel utilisation, imports, exports, and final consumption [33]. By using this data for one studied year, 2008, the magnitude of the uncertainties due to the above-mentioned simplification was analysed. The potential error in the annual consumption-based CO_2 emission intensity of electricity due to imports and related quarterly variation in the use of combustible fuels were calculated using equations (7) and (8).

In LCA (life cycle assessment), there are several ways of allocating emissions for various products in multi-product processes [34,35]. In determining GHG emissions of electricity in combined heat and power production, the allocation issue arises. CHP plants are built to jointly produce electricity and heat, which renders them economically competitive [30]. In general, the allocation of emissions can be based on a physical relationship, such as the energy or exergy content of the products, or on some other relationship, such as the price of the products [21]. Additionally, for CHP production, allocation methods based on fuel use in hypothetical alternative stand-alone production of heat, power, or both heat and power have been introduced (e.g.[36,37]). The method selected for the allocation procedure has a significant impact on the results. Frischknecht [30] used energy content, exergy content, price, ‘motivation heat’ and ‘motivation electricity’ as examples of parameters for determining the allocation factors for power and heat. As regards the emissions allocated to power, ‘motivation heat’ and ‘motivation power’ reflect the lower and upper limits, respectively, allocating 0% and 100% to the power. Both of these options can be reasonably used for allocation when either heat or power can be clearly assumed to be the main product. However, this is not usually the case, as both products typically have economic value. As power has higher exergy content and, normally, a significantly higher price level per energy unit produced compared to heat [30], the allocation of all emissions to heat can generally be considered misleading. Graus and Worrell [4] employed five different methods for calculating the CO_2 intensity of power generation. The lowest intensity was calculated by allocating emissions according to the energy output of heat and power in enthalpic terms, the highest by employing a ‘motivation power’ method in which all emissions are allocated to electricity. The results based on many other allocation factors, such as exergy content and product price, typically fall within this range [4,30]. Thus, we selected the allocation factor based on the energy content of heat and power outputs from CHP for the lowest limit. In this method, emissions are allocated on an equal basis to electricity and heat output in enthalpic terms (weighting factor $A = 0.5$ in equation (1)). For the upper limit of power-related CO_2 emissions from CHP we selected the ‘motivation electricity’ method (weighting factor $A = 1.0$ in equation (1)).

The annual national CO_2 emissions allocated to electricity production for each country were calculated using equation (1):

$$E_{\mathrm{el\,prod}} = E_{\mathrm{ep}} + E_{\mathrm{CHP}} * \left(\frac{A * \mathrm{el}_{\mathrm{CHP}}}{A * \mathrm{el}_{\mathrm{CHP}} + (1-A) * h_{\mathrm{CHP}}} \right) + E_{\mathrm{own}} * \frac{\mathrm{el}_{\mathrm{tot}}}{\mathrm{el}_{\mathrm{tot}} + h_{\mathrm{tot}}} + E_{\mathrm{autoel}} + E_{\mathrm{auto\,CHP}} * \frac{A * \mathrm{el}_{\mathrm{auto\,CHP}}}{A * \mathrm{el}_{\mathrm{auto\,CHP}} + (1-A) * h_{\mathrm{auto\,CHP}}} \quad (1)$$

in which

$E_{\mathrm{el\ prod}}$ = annual CO_2 emissions allocated to total electricity production

E_{ep} = annual CO_2 emissions from main activity electricity plants (excluding CHP plants and own use)

E_{CHP} = annual CO_2 emissions from main activity CHP plants

A = weighting factor for allocating CO_2 emissions between electricity and heat

$\mathrm{el}_{\mathrm{CHP}}$ = annual electricity output from main activity CHP plants

h_{CHP} = annual heat output from main activity CHP plants

E_{own} = annual CO_2 emissions from own use of electricity, CHP, and heat plants

$\mathrm{el}_{\mathrm{tot}}$ = annual total electricity output from electricity, CHP, and heat plants

h_{tot} = annual total heat output from electricity, CHP, and heat plants

$E_{\mathrm{auto\ el}}$ = annual CO_2 emissions from autoproducer electricity plants (excluding CHP plants and own use)

$E_{\mathrm{auto\ CHP}}$ = annual CO_2 emissions from autoproducer CHP plants

$\mathrm{el}_{\mathrm{auto\ CHP}}$ = annual electricity output from autoproducer CHP plants

$h_{\mathrm{auto\ CHP}}$ = annual heat output from autoproducer CHP plants

The annual electrical energy produced and transferred to final consumption points[1] within a country was calculated using equation (2):

$$\mathrm{el}_{\mathrm{pat}} = \mathrm{el}_{\mathrm{cons}} - \mathrm{el}_{\mathrm{imp}} + \mathrm{el}_{\mathrm{exp}} \quad (2)$$

in which

$\mathrm{el}_{\mathrm{pat}}$ = annual electrical energy produced and transferred to final consumption points within a country

$\mathrm{el}_{\mathrm{cons}}$ = annual total final electricity consumption [refers to electricity production plus imports minus exports minus electricity used at power stations (own use) minus electricity used for pumped storage, heat pumps, and electric boilers minus transmission and distribution losses minus energy industry consumption]

$\mathrm{el}_{\mathrm{imp}}$ = annual electricity imports (absolute value)

$\mathrm{el}_{\mathrm{exp}}$ = annual electricity exports (absolute value)

The annual national production-based CO_2 emission intensity of electricity (g CO_2/kWh$_e$) was calculated using equation (3):

$$I_{\mathrm{pb}} = \frac{E_{\mathrm{el\ prod}}}{\mathrm{el}_{\mathrm{pat}}} \quad (3)$$

The annual national CO_2 emissions embodied in electricity imports to a country were calculated using equation (4):

[1] Electricity exports from a producing country are considered as final consumption points of the particular country.

$$E_{\text{emb imp}} = \sum_{j=1}^{J} \text{el}_{\text{imp},j} * I_{\text{pb},j} \quad (4)$$

in which

$E_{\text{emb imp}}$ = annual CO_2 emissions embodied in electricity imports to a given country

j = index of country from which electricity is imported

J = number of countries from which electricity is imported

$\text{el}_{\text{imp},\,j}$ = annual electricity imports from country j

$I_{\text{pb},\,j}$ = annual production-based CO_2 emission intensity of electricity in country j

The annual national CO_2 emissions from electricity production consumed domestically were calculated using equation (5):

$$E_{\text{dom}} = I_{\text{pb}} * (\text{el}_{\text{pat}} - \text{el}_{\text{exp}}) \quad (5)$$

The annual national consumption-based CO_2 emission intensity of electricity (g CO_2/kWh$_e$) was calculated using equation (6):

$$I_{\text{cb}} = \frac{E_{\text{dom}} + E_{\text{emb imp}}}{\text{el}_{\text{cons}}} \quad (6)$$

The potential error (%) in the annual national production-based CO_2 emission intensity of electricity (g CO_2/kWh$_e$) due to quarterly differences in the use of combustible fuels was calculated using equation (7):

$$\Delta I_{\text{pb}} = \frac{\text{el}_{\text{CF},k}}{\text{el}_{\text{PROD},k}} * \frac{\text{el}_{\text{PROD},y}}{\text{el}_{\text{CF},y}} - 1 \quad (7)$$

in which

ΔI_{pb} = potential error (%) in the annual national production-based CO_2 emission intensity of electricity

$\text{el}_{\text{CF},\,k}$ = electricity output from combustible fuels in quarter k in year y

$\text{el}_{\text{PROD},\,k}$ = domestic production of electricity in quarter k in year y

$\text{el}_{\text{CF},\,y}$ = electricity output from combustible fuels in year y

$\text{el}_{\text{PROD},\,y}$ = domestic production of electricity in year y

The potential error (%) in the annual national consumption-based CO_2 emission intensity of electricity (g CO_2/kWh$_e$) due to quarterly differences in the use of combustible fuels was calculated using equation (8):

$$\Delta I_{\text{cb}} = \frac{\sum_{j=1}^{J} \left(\text{el}_{\text{imp},j} * I_{\text{pb},j} * \Delta I_{\text{pb},j} \right)}{\text{el}_{\text{cons}} * I_{\text{cb}}} \quad (8)$$

in which

ΔI_{cb} = the potential error (%) in the annual national consumption-based CO_2 emission intensity of electricity

j = index of country from which electricity is imported

J = number of countries from which electricity is imported

3. Results

The calculated CO_2 emissions from electricity production for the OECD countries combined were approximately 3.7–3.9 Gt in 1990 and 4.7–5.0 Gt in 2008. The annual production-based CO_2 emission intensity of electricity decreased steadily by approximately 10% between 1990 and 2008, from 579 to 612 g CO_2/kWh$_e$ in 1990 to 507–536 g CO_2/kWh$_e$ in 2008. The given ranges derive from the selected allocation method for CHP, with the lower end corresponding to the energy-based allocation and the higher end to the 'motivation electricity' method. The impact of the choice of allocation method was not very significant at the overall OECD level, as only around 10% of electricity was produced by CHP (Table S1 in the supplementary data). However, the impact was highly significant at the country level, for some countries, as later discussed.

The variation in annual production-based CO_2 emission intensities of electricity in the studied countries, was significantly high, ranging from almost zero in Norway during all the studied years to over 1800 g CO_2/kWh$_e$ in Poland in 1990 (Tables S2 and S3 in the supplementary data). However, high values of over 1000 g CO_2/kWh$_e$ occurred only in three countries, Poland, the Czech Republic and Greece, during the studied period. In these countries, the use of fossil fuels, in particular coal, constituted a significant proportion of electricity production. The high values may also indicate poor quality of the original data. Besides Norway, other examples of countries with low production-based CO_2 emission intensities were Sweden and Switzerland. The higher the fossil fuel-based electricity production was in a given country, the higher was the CO_2 emission intensity of energy production. The share of fossil fuels of the electricity production mix varied significantly between countries [6].

The annual variation in production-based CO_2 emission intensity of electricity was moderate at the average OECD level, but considerable for many individual countries due to changes in the fuel mix and in production technologies. Examples of such countries are Luxembourg, Norway, Finland, Sweden, Denmark, and France. For the Nordic countries, in particular, annual fluctuations in hydropower and nuclear power production significantly affected the respective amount of fuel used in electricity production.

The allocation procedure for CHP increased the variability of the results when the amount of electricity produced with CHP was high. Examples of countries with a relatively high share of CHP of electricity production are Poland, Denmark, Finland, and Sweden (Table S1 in the supplementary data). Relatively, the largest range in estimated production-based CO_2 emission intensity of electricity due to the allocation procedure for CHP was in Sweden, where the lower end (energy-based allocation) CO_2 emissions totalled only 30% of the CO_2 emissions in the higher end (all for electricity) on average between 2000 and 2008. Other countries where the respective ratio due to variation was significant were Switzerland (54%), Denmark (55%), Norway (57%), and Finland (65%). According to the statistics, CHP production plays a role in some countries without any heat output. A case example is Italy, where CHP production increased significantly during the 1990s, but its first year of heat output was 2004. Thus, the choice of allocation method does not affect the results prior to that year.

The difference between national production-based and consumption-based CO_2 emission intensity of electricity was highly significant for Switzerland, Norway, Slovakia, Austria, and Sweden, and fairly significant for Denmark, Finland, Hungary, and Italy (Fig. 1). Of these countries, only Denmark was a net exporter of CO_2 emissions embodied in electricity trade. This means that Denmark sold electricity with a lower CO_2 emission intensity than it purchased from other countries. For the other above-mentioned countries the opposite was true. For the rest of the studied countries, the difference was typically less than 10% within the studied years. The Netherlands, for example, imports a significant share of its final electricity consumption, but mainly from Germany, in which the CO_2 emission intensity of electricity production is relatively close to that of the Netherlands.

For a few European countries with a high share of electricity trade compared to final electricity consumption, the CO_2 emissions embodied in electricity trade were significant compared to overall national CO_2 emissions. Such countries include Switzerland, Slovakia, Luxembourg, Austria, and Finland (Fig. 2). Here again, the impact of the allocation procedure was considerable in cases where a significant amount of the electricity production of the country

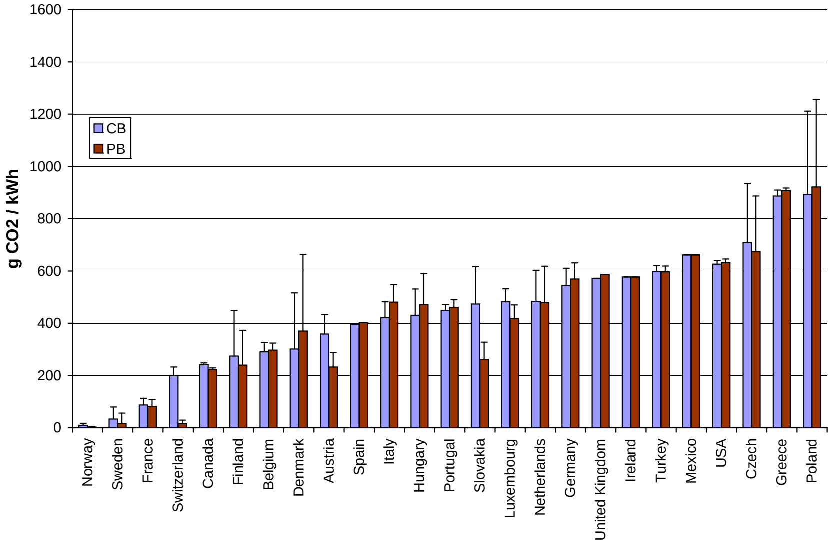


Fig. 1. Production-based (PB) and consumption-based (CB) CO_2 emission intensities of electricity (g CO_2/kWh$_e$) in OECD countries with electricity trade averaged between 2006 and 2008. The error bars illustrate the impact of the selected method to allocate CO_2 emissions between electricity and heat in combined heat and power production (CHP). The coloured columns correspond to the energy-based allocation and the upper limit of the error bars correspond to the 'motivation electricity' method.

from which the electricity was imported to the studied country was based on CHP production. However, for the majority of OECD countries, the electricity trade had an insignificant impact on overall CO_2 emissions. This is mainly due to the low amount of electricity traded compared to final electricity consumption.

The potential impact of averaging trade over a year instead of a shorter time period did not have a significant impact on the annual consumption-based CO_2 emission intensity of electricity. The difference between using annual data and quarterly data was estimated to be less than ±10% for each of the OECD countries in 2008 (Table S8 in the supplementary data). The impact was the higher the more a country imported electricity from a country in which the variation in the use of combustible fuels in electricity production was relatively high. The highest difference was found

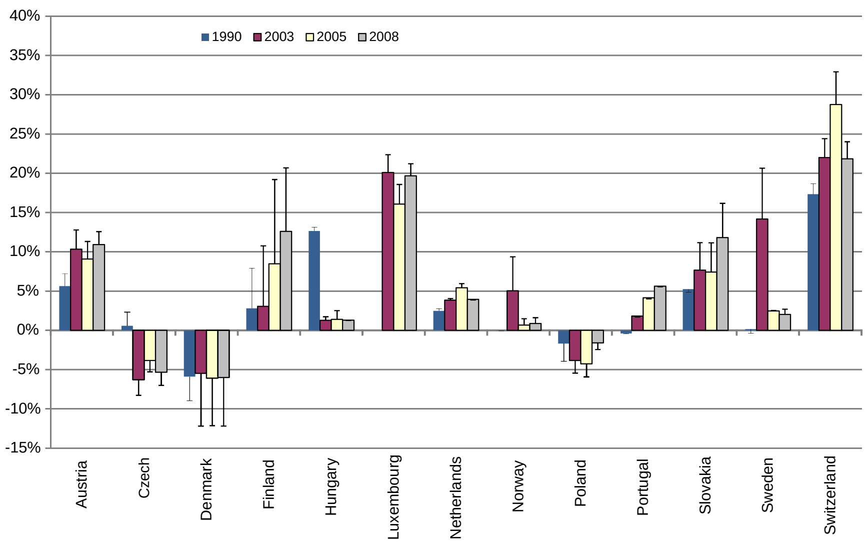


Fig. 2. CO_2 emissions embodied in net imports of electricity compared to total national CO_2 emissions (excl. LULUCF) for countries in which the share exceeds 5% for any studied year (negative values refer to export of embodied CO_2 emissions). The error bars illustrate the impact of the selected method to allocate CO_2 emissions between electricity and heat in combined heat and power production (CHP). The coloured columns correspond to the energy-based allocation and the ends of the error bars correspond to the 'motivation electricity' method.

for Hungary (9%), Greece (8%), and Finland (7%). Most of the uncertainty resulted from imports from Ukraine, Bulgaria, and Russia. For these origins, the quarterly variation in CO_2 emission intensity of electricity was not known.

4. Discussion and conclusions

4.1. Differences in production-based and consumption-based CO_2 emission intensities of electricity

For several OECD countries, the production-based and consumption-based CO_2 emissions of final electricity consumption deviated significantly. This is true when there is a difference in the CO_2 emission intensities of produced and transferred electricity between the country considered and the countries from which the electricity is imported to the country considered. Thus, a country may have significant amount of CO_2 emissions embodied in its imports or exports even if the net electricity trading of the country is at a low level. Switzerland, Luxembourg, Norway, and Sweden had the highest CO_2 emissions embodied in imports in relative terms. In Switzerland, the production-based CO_2 emissions of electricity were low, as nearly all electricity was produced with hydro and nuclear power. Switzerland's imports constituted more than half of its consumption; however, its exports were of the same order of magnitude as its imports. The electricity imported from France and, in particular, from Germany had much higher intensity than the Swiss exports. Luxembourg imported nearly as much as it consumed. Luxembourg is a small country with primarily natural gas-based domestic production, and its CO_2 emission intensity was relatively low. In Norway and Sweden the CO_2 emission intensity of electricity production was low due to significant use of hydropower in both countries and also nuclear power in Sweden.

Denmark, the Czech Republic, and Poland had more CO_2 emissions embodied in electricity exports than in their imports. Thus, their production-based CO_2 emissions were greater than those of their own consumption. In Denmark, 80% of total production in 2008 was produced with fuels, mainly hard coal and natural gas. Even though Denmark imported slightly more electricity than it exported in 2008, it had more CO_2 emissions embodied in exports than imports. Denmark imported low CO_2 emission intensive electricity from Sweden and Norway. The Czech Republic had relatively high production-based CO_2 emissions due to significant coal use. Exports from the Czech Republic were more than twice as large as imports to the country, and around one fourth of the annual production in 2008 was exported. Poland had very high CO_2 emissions embodied in exports, as its production-based CO_2 emissions were the highest in the OECD due to significant use of coal. Poland's trade in 2008 was at a low level relative to total consumption, although its exports were slightly higher than imports.

4.2. Uncertainties

Our quantitative uncertainty analysis showed that ignoring the exact moment of the electricity trade, by considering only annual averaged data, probably does not have a significant impact on the consumption-based CO_2 emission intensity of electricity in various countries. Naturally, the calculated impact could have been somewhat higher if monthly or weekly data would have been applied. One clear limitation in our approach to determining the CO_2 emissions of electricity trade is the assumption that the production mix of the electricity traded from a country corresponds to the average production mix of the electricity produced within the country. This is not necessarily the case in practice. The average electricity production mix of a country consists of a number of production mixes of smaller regions. Thus, the electricity that is traded from a region of a country to another country should correspond to the production mix of that particular region, taking into account inland transfers. However, how to determine the appropriate size of region in this context is not clear. In addition, comparable public statistics for electricity production, consumption, and transfers are not available for a regional analysis. More research and agreements of various stakeholders are likely required in order to determine more specific data on the GHG emissions embodied in electricity trade.

Uncertainty in this study is also due to the data used. There is some uncertainty related to the accounting of CO_2 emissions from fossil fuel combustion due to problems in determination of fuel-specific characteristics such as moisture, lower heating value, and carbon content (e.g.[38]). In addition, the figures related to CHP should be interpreted with caution. The dividing line between main producers and autoproducers and between inputs and outputs of the CHP plants is unclear, and not always consistent. It is likely that the impact of these uncertainties on our results is not significant, but we could not analyse the magnitude quantitatively in this paper.

For six countries, namely Germany, Poland, the Czech Republic, Slovakia, Spain, and Turkey, some electricity imports from non-specified origins were identified. For Poland and Slovakia 100% of the electricity imported in 1990 was from non-specified origins. The respective figures for the other four countries were: Germany 21% (1990), Czech Republic 6–20% (2003–2008), Spain 1% (2005), and Turkey 1–8% (2000–2002, 2008). The share of imports of final electricity consumption in 1990 was 10% and 30% for Poland and Slovakia, respectively. For the other four countries the corresponding share was insignificant. Consequently, the consumption-based CO_2 emission intensity of electricity was highly uncertain due to imports from non-specified origins only for Slovakia in 1990. CO_2 emission intensity of electricity was not available for Luxembourg between 1990 and 2001. However, this had no impact on any other countries' consumption-based CO_2 emission intensity of electricity, as the electricity was only exported to Germany, corresponding less than 0.2% of the final electricity consumption of Germany.

4.3. GHG emission leakage

Most OECD countries involved in electricity trading are a part of the European Union. The majority of CO_2 emissions from fuel combustion are regulated and monitored at the EU level under the EU Emission Trading Scheme (ETS) from 2005 onwards [39]. Of the non-EU countries studied here, Norway is included in the system. Electricity trading between the countries included in the EU ETS often induces some GHG emission leakage, but within the EU ETS, the overall emissions are limited by an annual cap. The CO_2 emissions from electricity production may induce leakage out of the EU ETS if electricity imports to the EU from the countries outside the EU ETS increase. Total emissions might also potentially increase due to more ineffective production in the countries to which emissions are shifted compared to the EU ETS region.

In total, imports from outside the EU ETS accounted for 2–3% of the final electricity consumption between 2000 and 2008 in the countries inside the EU ETS included in the study. Countries with significant imports from outside the EU ETS include Finland with 16%, Hungary with 13%, Greece with 10%, and Italy with 9% imports of the total final electricity consumption in 2008. The share of the imports of final electricity consumption slightly increased only in Finland since the introduction of the EU ETS in 2005.

Although electricity trade seems to constitute a relatively minor part in the overall CO_2 emissions embodied in trade, it is a factor to be considered. In addition, electricity trading is likely becoming an increasingly important factor in the future [8,9]. Therefore, the extension of the EU ETS to countries with significant electricity exports to the EU ETS region could offer a means of avoiding the unintended leakage effect. Another option could be the introduction of consumption-based method to determine the emissions for the electricity imported to the EU ETS region. An emission regulation system similar to the EU ETS effectively avoiding significant emission leakage could work also for other electricity market areas with significant electricity trade between nations.

4.4. Consumption-based GHG performance rules for products

The annual average production-based CO_2 emission intensity of electricity (Tables S2 and S3 in the supplementary data) is often used in basic life cycle assessment. This may be highly misleading with respect to certain countries. The consumption-based estimates presented in this paper with respect to the CO_2 emissions embodied in the electricity trade provide a more reliable picture of the GHG emission intensity of the electricity consumed (Tables S4 and S5 in the supplementary data). Consequently, we advocate the use of the consumption-based method in preference to the production-based method for LCA purposes.

Allocation of the emissions for power and heat in CHP can have significant impacts on the national GHG emission intensities of electricity consumption, as shown in this paper. Allocation is one of the most problematic methodological issues in LCA, and the allocation procedure is always a subjective choice. This problem needs to be addressed when introducing GHG performance rules for products. We selected allocation methods that gave a reasonable range for the purposes of this study. We are unable, however, to suggest any allocation method as being superior to others based on the results of this study.

Our figures do not include upstream GHG emissions. These, however, typically constitute a relatively low share of GHG emissions of the overall electricity production mix (e.g.[40,41,42]), although for certain power production technologies they may be significant [26,27]. However, an extensive shift in energy production systems may occur within the next few decades with the large-scale introduction of low GHG emission intensive power production technologies as a result of ambitious climate change mitigation targets [1]. Consequently, in the overall life cycle of electricity consumption, the contribution of GHG emissions not related to direct fuel combustion might increase significantly and would therefore need to be considered more carefully. In particular, GHG emissions related to the cultivation and harvesting of bioenergy has already been widely discussed (e.g.[43,44]).

Other critical methodological issues related to GHG emissions from electricity consumption also need to be considered. Various geographical choices including regional, country, and market area levels as well as temporal choices including instant, monthly, annual, and perennial considerations, can be rationalised. Soimakallio et al. [37] concluded that national or regional production mix figures should only be used for analyses concerning electricity consumption at the national or regional level, respectively. They also concluded that a solution for avoiding arbitrary selection of electricity market area could be the introduction of figures based on the contract between the electricity seller and the customer with real-time accounting. Currently, such data and respective reporting practices do not generally exist. One important research question is the definition of appropriately short time periods for the determination of GHG emission intensity figures.

4.5. Concluding remarks

The CO_2 emissions attributable to electricity consumption vary country-specifically and annually. They also vary depending on the method of analysis, such as the emissions allocation method used for combined heat and power production, and the consideration of electricity trading. Use of the production-based method for determining the emissions of electricity consumption within countries for purposes of assessing, verifying, and monitoring the GHG performance of specific products may be highly misleading. The consumption-based method should therefore be preferred. Uncertainties can be reduced by improving the quality of the data. Regarding GHG performance rules for products, especially mandatory, open methodological issues, such as allocation procedure, need to be solved keeping in mind that any single solution is always subjective. According to our results, the absolute value of CO_2 emissions embodied in electricity net imports may be relatively significant for some countries. Regarding the overall CO_2 emissions embodied in trade, electricity plays a relatively minor role. However, unless effective emission reduction and regulation measures are implemented more intensively, this role may increase in the future.

Climate policy instruments, such as national GHG emission reduction targets and product GHG performance rules, may act as required incentives in climate change mitigation. However, if implemented as incomplete and applied inappropriately, such instruments entail serious risks with respect to GHG emission leakage. Effective emission regulation should result in the intended emission reductions. The emission accounting methods used in regulation should, therefore, be fit for task. It should be noted that assessment of the environmental impacts of various decisions, such as climate policies and measures, based on retrospective approaches should be carefully assessed using prospective consequential assessment methods. Changes in electricity consumption have short- and long-term impacts which may vary significantly from those assessed using retrospective average figures, such as the ones presented in this paper [37]. The obvious advantage of the retrospective approach is the ability to reliably verify and monitor GHG performance, which is crucial for effective emission regulation.

Acknowledgements

We thank VTT Technical Research Centre of Finland and Helsinki University Centre for Environment HENVI for funding. We are grateful also to the anonymous reviewers for their valuable comments on this paper.

Appendix. Supplementary data

Supplementary data related to this article can be found online at doi:10.1016/j.energy.2011.12.048.

References

[1] The Intergovernmental Panel on Climate Change (IPCC). Climate change 2007 WGIII. IPCC; 2007.
[2] International Energy Agency (IEA). Energy technology perspectives. Paris: IEA; 2010.
[3] International Energy Agency (IEA). CO_2 emissions from fuel combustion database. Paris: IEA; 2010.
[4] Graus W, Worrel E. Methods for calculating CO_2 intensity of power generation and consumption: a global perspective. Energ Policy 2011;39:613–27.
[5] International Energy Agency (IEA). Energy technology perspectives 2008-scenarios & strategies to 2050. Paris: IEA; 2008.
[6] International Energy Agency (IEA). Energy balances database. Paris: IEA; 2010.
[7] International Energy Agency (IEA). Electricity information. Paris: IEA; 2010.
[8] European Network of Transmission Systems Operations for Electricity. Ten-year network development plan 2010–2020. Brussels, Belgium: Entso-e; 2010.
[9] PriceWaterHouseCoopers (PWC). 100% renewable electricity. a roadmap to 2050 for Europe and North Africa, www.pwc.com/sustainability; 2010.

[10] Ekvall T, Weidema BP. System boundaries and input data in consequential life cycle inventory analysis. Int J LCA 2004;9:161–71.
[11] Lund H, Vad Mathiensen B, Christensen J, Schmidt JH. Energy system analysis of marginal electricity supply in consequential LCA. Int J LCA 2009;15:260–71.
[12] Pehnt M, Oeser M, Swider DJ. Consequential environmental system analysis of expected offshore wind electricity production in Germany. Energy 2008;33: 747–59.
[13] Ekholm T, Soimakallio S, Moltmann S, Höhne N, Syri S, Savolainen I. Effort sharing in ambitious global climate change mitigation scenarios. Energ Policy 2009;38:1797–810.
[14] Searchinger T, Heimlich R, Houghton RA, Dong F, Elobeid A, Fabiosa J, et al. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 2008;319:1238.
[15] Nijkamp P, Wang S, Kremers H. Modeling the impacts of international climate change policies in a CGE context: the use of the GTAP-E model. Econ Model 2005;22:955–74.
[16] United Nations Framework Convention on Climate Change (UNFCCC). Kyoto protocol. Kyoto: UNFCCC; 1997.
[17] United Nations Framework Convention on Climate Change (UNFCCC), Copenhagen accord (http://unfccc.int/files/meetings/cop_15/application/pdf/cop15_cph_auv.pdf).
[18] Peters G, Hertwich E. CO_2 embodied in international trade with implications for global climate policy. Environ Sci Technol 2008;42:1401–7.
[19] Davis SJ, Caldeira K. Consumption-based accounting of CO_2 emissions. PNAS 2010;107:5687–92.
[20] Peters G, Minx J, Weber C, Edenhofer O. Growth in emission transfers via international trade from 1990–2008. PNAS 2011;108:8903–8.
[21] International Organization for Standardization. ISO 14040–14044:2006, http://www.iso.org; 10 October 2010.
[22] The greenhouse gas protocol initiative, http://www.ghgprotocol.org/; 10 October 2010.
[23] BSI Standards Solutions. PAS 2050, http://shop.bsigroup.com/; 10 October 2010.
[24] Environmental product declaration (EPD), http://www.environdec.com/; 10 October 2010.
[25] 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. Off J Eur Union 5.6.2009.
[26] Frischknecht R, Althaus HJ, Bauer C, Doka G, Heck T, Jungbluth N, et al. The environmental relevance of capital goods in life cycle assessments of products and services. Int J LCA 2007;11 [online first].
[27] Weisser D. A guide to life-cycle greenhouse gas (GHG) emissions from electric supply technologies. Energy 2007;32:1543–59.
[28] U.S. Environmental Protection Agency. Emissions&generation resource integrated database. eGRID, http://www.epa.gov/cleanenergy/egrid/index.htm; 2007.
[29] Narayanan BG, Walmsley TL. Global trade, assistance, and production: the GTAP 7 data base (center for global trade analysis. West Lafayette, IN: Purdue University; 2008.
[30] Frischknecht R. Allocation in life cycle inventory analysis for joint production. Int J LCA 2000;5:85–95.
[31] Weber CL, Jaramillo P, Marriott J, Samaras C. Life cycle assessment and grid electricity: what do we know and what can we know? Environ Sci Technol 2010;44:1895–901.
[32] United Nations Framework Convention on Climate Change (UNFCCC), Greenhouse gas inventory data, http://unfccc.int/ghg_data/items/3800.php.
[33] International Energy Agency (IEA). Oil, gas, coal & electricity, quarterly statistics, fourth quarter. Paris: IEA; 2009.
[34] Ekvall T, Finnveden G. Allocation in ISO 14041-a critical review. J Cleaner Prod 2001;9:197–208.
[35] Finnveden G, Hauschild MZ, Ekvall T, Guinée J, Heijungs R, Hellweg S, et al. Recent developments in life cycle assessment. J Environ Manage 2009;91: 1–21.
[36] The World Bank. Regulation of heat and electricity produced in combined-heat-and-power plants. World Bank Technical Paper No; October 6, 2003.
[37] Soimakallio S, Kiviluoma J, Saikku L. The complexity and challenges of determining GHG emissions from grid electricity consumption and conservation in LCA - a methodological review. Energy 2011;36:6705–13.
[38] Marland G. Uncertainties in accounting for CO_2 from fossil fuels. J Ind Ecol 2009;12:136–9.
[39] Directive 2009/29/EC of the European parliament and of the council of 23 april 2009 amending directive 2003/87/EC so as to improve and extend the greenhouse gas emission allowance trading scheme of the community. Off J Eur Union 5.6.2009.
[40] Kim S, Dale BE. Life cycle inventory information of the United States electricity system. Int J LCA 2004;10:294–304.
[41] Santoyo-Castelazo E, Gujpa H, Azapagic A. Life cycle assessment of electricity generation in Mexico. structural analysis of electricity consumption by productive sectors. the Spanish case. Energy 2011;36:1488–99.
[42] Lee KM, Lee SY, Hur T. Life cycle inventory analysis for electricity in Korea. Energy 2004;29:87–101.
[43] Repo A, Tuomi M, Liski J. Indirect carbon dioxide emissions from producing bioenergy from forest harvest residues. Glob Change Biol 2010;3:107–15.
[44] Fargione J, Hill J, Tilman D, Polasky S, Hawthorne P. Land clearing and the biofuel carbon debt. Science 2008;29:1235–8.

PAPER V

Top-down approaches for sharing GHG emission reductions

Uncertainties and sensitivities in the 27 European Union Member States

In: Environmental Science & Policy 11(8), 723–734.

ENVIRONMENTAL SCIENCE & POLICY 11 (2008) 723–734

available at www.sciencedirect.com

ScienceDirect

journal homepage: www.elsevier.com/locate/envsci

Top-down approaches for sharing GHG emission reductions: uncertainties and sensitivities in the 27 European Union Member States

Laura Saikku [a,*], *Sampo Soimakallio* [b]

[a] *University of Helsinki, Department of Biological and Environmental Sciences, P.O. Box 27, 00014 University of Helsinki, Finland*
[b] *VTT Technical Research Centre of Finland, P.O. Box 1000, FIN-02044 VTT, Finland*

ARTICLE INFO

Published on line 5 September 2008

Keywords:
GHG emissions
Effort sharing
EU27
GHG intensity of economy
Top-down approach

ABSTRACT

To reduce GHG emissions, the 27 European Union Member States committed themselves in 2007 to reduce emissions from 1990 levels by 20% by 2020. In January 2008, the EU Commission gave the first country-specific proposals to reduce emissions in sectors outside the EU emission trading system (non-ETS). In this study, we looked at several ways of sharing emission reductions in the non-ETS sector. We considered population and economic growth as significant drivers of the development of emissions. In particular, we analyzed development in GHG intensity of economies. Reduction requirements vary greatly among countries depending on the principle of effort sharing. The results of our calculations can be perceived as examples of how effort sharing between the EU Member States could look like when certain assumptions are made. Generally they illustrate the sensitivity of the results to data used, assumptions made, and method applied. The main strength of simple top-down approaches is transparency. A major weakness is a very limited ability to consider national circumstances. Political negotiations are ultimately crucial; an analysis like this provides material for negotiations and makes a contribution to solving the effort-sharing problem. As future development is partly unpredictable, implementation of some kind of subsequent adjustment could be considered during the process.

1. Introduction

Human-inflicted greenhouse gas emissions affect the global temperature. The global mean temperature is expected to increase significantly and there is a growing risk of extreme climatic events (IPCC, 2007). In 1996, the European Commission recommended that the rise in global average temperature should be limited to 2 °C above the pre-industrial level. The European Union, accounting for approximately 15% of global GHG emissions (IEA, 2006), is in the forefront of combating climate change. In March 2007, the EU Prime Ministers agreed on a post-Kyoto target, a commitment of a 20% reduction of GHG emissions by 2020 from 1990 levels. On the condition that other countries also commit to reductions, they agreed that the EU countries should reduce GHG emissions by 30% for the same period.

To arrive at 20% emission cuts by 2020 is, however, challenging. To achieve the particular target of reducing CO_2 emissions alone is a demanding task, as this would mean around a two-fold improvement in the decarbonisation and dematerialisation rates occurred in the 27 member states currently comprising the European Union (EU27) during

* *Corresponding author*. Tel.: +358 9 191 57769.
E-mail address: laura.saikku@helsinki.fi (L. Saikku).
1462-9011/$ – see front matter
doi:10.1016/j.envsci.2008.07.002

1993–2004 (Saikku et al., 2008). In that period, affluence and population grew strongly, more than offsetting the modest efficiency gains. The rates of dematerialisation and decarbonisation varied between countries, with faster development in the 12 new Member States than in the earlier EU15 members. Consideration of reducing non-CO_2 greenhouse gas emissions provides some relief to the challenge as there are many cost-effective options to reduce these emissions (Delhotal et al., 2006). Consideration of ecosystem biomass as a carbon sink, as well as implementation of emission reductions in developing countries through the Clean Development Mechanism (CDM) would have considerable significance for the EU, but these are dependent on the development of the international climate policy.

The European Commission proposed legally binding post-Kyoto targets for its current 27 Member States on 23 January 2008 (European Commission, 2008). Emissions would be reduced separately in the emission trading sector (including mainly energy and industrial GHG emissions), and sectors not included in emission trading (non-ETS), such as residential, agriculture, transportation and waste management. The proposed reduction in the Emission trading sector was 21% and in the non-ETS sector 10% from the 2005 level (European Commission, 2008). The emission trading sector will be administered at the EU level whereas the other sectors will be given an overall national target. The proposal in the non-ETS sector divides emission reduction efforts between Member States based on simple GDP per capita criteria. According to the Commission, countries with low GDP/capita and high GDP growth expectations should be allowed to increase their emissions.

Effort-sharing approaches can be studied from many different perspectives. Besides top-down methods, an approach may be based on more sophisticated and data-oriented bottom-up methods. For EU Member States, the internal burden sharing of the Kyoto Protocol was previously negotiated on the basis of Triptych method (Blok et al., 1997). Triptych is a relatively simple sectoral approach for sharing national emission allowances, serving to improve understanding about differences in national circumstances relevant to burden differentiation. The approach enhances population size and population growth, economic structure, emission intensity of economy as CO_2/GDP, affluence as GDP/capita, standard of living as CO_2/capita, energy efficiency as CO_2/Energy, fuel mix and climate as heating degree days (Phylipsen et al., 1998). Triptych has been later expanded as a global application to set post-Kyoto targets. The sensitivity and suitability of the extended Triptych approach developed by Ecofys (Phylipsen et al., 2004) to set emission quotas was tested and analyzed by Soimakallio et al. (2006). In addition to the Triptych approach, other options for differentiating GHG mitigation commitments internationally and in the EU, particularly for the post-Kyoto period, were summarised, among others, by Sijm et al. (2007).

Any effort-sharing principle should be politically acceptable with respect to fairness principles and operational requirements (Torvanger and Ringius, 2001). The key issue with an effort-sharing method is the dilemma between its transparency, on the one hand, and its ability to take into account national circumstances, on the other hand (Soimakallio et al., 2006). The data used for calculating the targets for the parties should be robust, generally acceptable, and transparent so as to be open to critical analysis. The latter requirement can easily conflict with effort-sharing methods based on sophisticated model calculations, which are typically required for responding to certain scopes of fairness. The strengths of simple top-down methods are relatively good availability and a limited amount of required data, as well as transparency of the method. However, the restricted ability to consider national circumstances and factors explaining top-down figures can be seen as a major weakness.

This paper presents a few top-down approaches to sharing the effort to reduce greenhouse gas emissions outside the EU's emission trading system within the EU countries by 2020. The top-down approaches studied are based on the economy's greenhouse gas intensity by taking into account the forecasted economic and population growth. In addition, we consider and evaluate the EU Commission preliminary proposal for effort sharing from the viewpoint of our results. Finally, we discuss different top-down approaches, sensitivities in the results and uncertainties related to studies based on forecasts.

2. Data sources and methods

2.1. *Scenarios in this study*

Top-down macro figures are used in the approaches studied to set the emission reduction targets for the Member States. The approach takes into account the current level of greenhouse gas emissions (Appendix A) and the forecasted growth of population and the economy in the different Member States, and simply sets the targets for greenhouse gas intensity of economy in terms of GHG/GDP for the non-ETS sector by applying various rules.

Four different effort-sharing scenarios were calculated for non-ETS emission reduction. The reduction is assumed to start in 2008. The total reduction in the non-ETS sector is determined through reductions in the ETS sector. ETS emissions in countries are reduced by 20% from 2005 verified emissions. The reduction in the ETS sector is determined by the grandfathering principle, where each country reduces their emissions by the same share. Non-ETS sector as a whole reduces 8% from 2005 level. Emission data for year 2005 is used as a starting point for reduction, assuming that 2005 emissions are equal to 2008, as 2005 is the latest year of available data. The GDP forecast for 2008 is used. All scenarios refer to non-ETS only.

Scenario 1: The annual rate of change in GHG/GDP is the same in all Member States during 13 years, 2008–2020.
Scenario 2: GHG/GDP becomes equal in all countries in 2020.
Scenario 3: National rates of GHG/GDP are the same as they were in 1993–2005. In order to reach a reduction of 20% by 2020, an additional reduction is required. This additional annual reduction is set constant

over time and the same for all countries in percentage terms.
Scenario 4: GHG per capita becomes equal in all countries in 2020.

2.2. Data

The historical data for greenhouse gas emissions and GDP, as well as forecasts for population growth (baseline variant by 2020) in the different Member States, were derived from the Eurostat database (2008). Verified ETS emissions for 2005 are from CITL (17 October 2007) database. Population growth is predicted to influence development to some extent, around 1%/year in Ireland and some small countries like Cyprus (Appendix B). Population acts as a downward force for, e.g. Bulgaria, Latvia, Lithuania and Estonia.

Forecasts of economic development were carried out according to a model described in more detail in Saikku et al. (2008). In the model, real GDP growth rates for 2007–2008 as reported by Eurostat (2008) were used in the forecasts. For projections of the development of total GDP after 2008, countries were divided into four groups based on the level of their affluence (GDP/capita) in 2006. The GDP's of the countries in the richest group are set to grow at a rate of 2%/year. The other three groups of countries converge to the average affluence level of the richest group at differing time-spans, depending on their initial level of affluence.

We used estimates for the non-ETS sectors' GDP in our calculation. The approximated GDP share of the Emission Trading Scheme (ETS) sectors is roughly based on Eurostat (2008) GDP data, on GDP of the energy industries, the manufacturing industries and construction, and industrial processes. The non-ETS sector GDP is a complement of the ETS sector GDP.

Non-ETS GDP growth in years 2008–2020 is expected to be considerable, more than 5%/year, for a few countries: Romania, Poland, Bulgaria, Hungary and Slovakia (Appendix B). GDP growth is projected to be most modest, around 2%/year, in some other western countries, for instance Germany and France.

We also compared required GHG intensities in our scenarios to recent historical development. Historical development in GHG/GDP during 1993–2005 was calculated for total GDP. Non-ETS GHG estimated for 1993 is based on Eurostat emissions for the energy industries, and manufacturing and industrial processes. GDP (PPP-corrected) for 1993 from Penn World Table (Heston et al., 2007).

2.3. Sensitivity analysis

We conducted the following test runs for all scenarios to analyze certain sensitivities involved in the results. In comparison to the base case presented above:

Test run 1: The base year for emissions is changed to 2004.
Test run 2: Emissions in the ETS sector are reduced by 20% from the second national allocation plans for 2008–2012, approved by the European Commission (European Commission, 2007a).
Test run 3: Emissions in the ETS sector are reduced 0% from the verified emissions in 2005.
Test run 4: GDP forecasts presented in Mantzos et al. (2003) and POLES model (Russ et al., 2007).[1]
Test run 5: The base year for GDP is changed to 2004 and 2005, in addition, overall GDP is used instead of non-ETS GDP.
Test run 6: Population forecasts are calculated according to Eurostat High and Low variants.

3. Results

The effort-sharing approaches studied varied relatively significantly in terms of greenhouse gas targets for 2020 in the non-ETS sector for EU Member States (Fig. 1, see detailed results for all countries in four scenarios in Appendix C). Countries' reduction targets are determined by their level of GHG emission in the starting year (2008), their current GDP and population level and growth expectations. Also historical development in GHG/GDP has an impact in one scenario.

In Scenario 1, all countries need to improve their GHG intensity of economy at the same rate. The emission reduction target depends on the growth rate of GDP. Those countries with highest estimated GDP growth are allowed to increase their emissions. The other way around, for example Germany has lowest expected GDP growth and tightest emission reduction target.

Scenario 2 assumes equal emission per GDP for all countries in 2020. The emission reduction target depends on the level of GHG/GDP in the starting year in relation to estimated GDP growth. Those countries with low GHG/GDP level in the starting year (2008) together with relatively high increase in expected GDP growth can emit the most (like Malta and Latvia). Sweden, in particular, is allowed to grow its emissions because the level of GHG/GDP in base year is low although its GDP growth is below the EU average.

Scenario 3 is based on historical rates of GHG/GDP. Emission reduction targets depend on historical rates of GHG/GDP multiplied with expected GDP growth. Those countries whose historical rate in GHG/GDP has been decreasing intensively combined with moderate GDP growth expectations get the toughest targets (like Ireland, Finland, United Kingdom, Denmark and Sweden). Although expected GDP growth in Latvia exceeds the EU average the country should reduce emissions significantly in Scenario 3 due to a remarkable decrease in historical GHG/GDP.

[1] The two reference forecasts of POLES and Mantzos et al. are more sophisticated approaches on GDP growth and are based on detailed system models. The growth expectations for several Eastern European countries are much more modest (max 3%/year for any EU country) compared to Saikku et al. In fact, some countries' GDP is expected even to decrease (Latvia in POLES and Mantzos et al., in addition, Bulgaria, Czech, Estonia, Hungary, Romania, Slovakia and Slovenia in Mantzos et al.). The forecast of POLES is PPP-corrected, similarly to Saikku et al. (2008) and the expected growth fall very close to each other for a few Western European countries (e.g. Denmark, Germany, Netherlands, Luxembourg, Finland, Belgium). Also, the growth expectations for these countries are more modest when comparing to Mantzos et al.

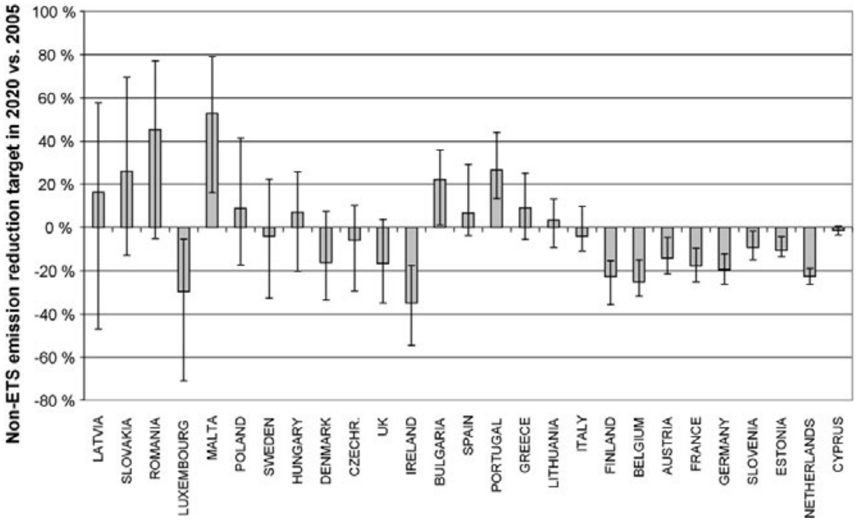


Fig. 1 – Average change in non-ETS emissions in different scenarios for 2020 in comparison with 2005. Error bars represent the variation range (min and max) in terms of percentage points. Countries furthest left have largest variation between scenarios.

Scenario 4 assumes equal emissions per capita for all countries in 2020. Emission targets depend on the factor determined by GHG/capita in the starting year in relation to population growth. Those countries with a low GHG/capita level in the starting year (2008) and/or high estimated increase in population growth are allowed to emit the most, like Malta, Latvia, Romania and Bulgaria.

Variation in emission targets between scenarios for particular countries was considerable for some of the Member States but more moderate for others (Fig. 1). The variation between scenarios was moderate when the determining factors in a particular country were close to the EU average in all scenarios. However, as the determining factors, e.g. GDP growth in Scenario 1 or emissions per capita in relation to population growth in Scenario 4, vary between scenarios, there is no clear answer why the deviation is more considerable for some countries than the others. The variation in terms of percentage points was large for Latvia, Slovakia, Romania and Luxembourg. For Cyprus, Netherlands and Estonia, and also, Slovenia, Germany and France, variation was small. The reduction targets proposed by the Commission (Appendix C) fall in the range of the results in our study for all countries, except Slovenia.

There is variation in the stringency of targets between scenarios for each country (Table 1). For example, Scenario 1 (equal non-ETS reduction in GHG/GDP), results in toughest targets for a few countries that are below the EU average both in 2005 and 2020, in terms of absolute GHG intensity of economy. Between different scenarios, Equal non-ETS GHG per GDP favours wealthy western countries like Denmark and France, but also Latvia. However, when other GDP forecasts are used, the GHG/capita option becomes more favourable for Latvia.

Scenario 3, historical rates of GHG/GDP, yields tougher targets compared to other scenarios for a few Western Europe countries like Ireland and some Eastern European countries, like Latvia and Lithuania. In Eastern Europe, a structural change occurred during 1993–2005 and more of the decrease in non-ETS GHG/GDP can be attributed to strong GDP growth than falling emissions. Also, in these countries, GHG/GDP in 1993 was at a high absolute level. In Ireland the absolute level was much above the EU average in the beginning of the historical period in comparison. The variation in terms of reduction targets is widest in Scenario 4; however, it is the most extreme only for a few countries. Looking at all countries' average requirements in different scenarios (Fig. 1), would result in −8% total change from 2005 level,

Table 1 – Toughest and easiest scenarios for countries

	Toughest	"Easiest"
Scenario 1	Austria, Italy, Portugal, Spain, Malta	Bulgaria, Estonia, Ireland, Poland, Romania, Lithuania
Scenario 2	Bulgaria, Czech Republic, Estonia, Hungary, Poland, Slovakia, Romania	Denmark, France, Latvia, Netherlands, Sweden, UK
Scenario 3	Cyprus, Denmark, Finland, France, Germany, Ireland, Latvia, Lithuania, Slovenia, Sweden, UK	Austria, Belgium, Czech Republic, Greece, Hungary, Italy, Luxembourg, Portugal, Slovakia, Spain
Scenario 4	Belgium, Greece, Luxembourg, Netherlands	Cyprus, Finland, Germany, Malta, Slovenia

Table 2 – Average annual non-ETS GHG/GDP intensity change requirement during 2008–2020 (13 years) in four scenarios, according to the EU proposal and historical change during 1993–2005

Member State	1993–2005 (%)	EU proposal (%)	SCE1 (%)	SCE2 (%)	SCE3 (%)	SCE4 (%)	AVERAGE in SCE1–4 (%)
Austria	0.6	−3.5	−3.7	−2.8	−2.3	−3.6	−3.1
Belgium	−0.2	−3.3	−3.7	−4.7	−3.1	−4.7	−4.1
Bulgaria	−1.7	−5.0	−3.7	−5.9	−4.6	−4.0	−4.5
Cyprus	−1.0	−4.3	−3.7	−3.7	−3.9	−3.6	−3.7
Czech Republic	−0.5	−3.8	−3.7	−6.6	−3.4	−5.0	−4.7
Denmark	−2.1	−3.9	−3.7	−1.4	−4.9	−3.3	−3.3
Estonia	−1.4	−2.8	−3.7	−4.5	−4.3	−4.4	−4.2
Finland	−2.4	−3.6	−3.7	−3.7	−5.3	−3.3	−4.0
France	−1.1	−3.2	−3.7	−2.6	−4.0	−3.0	−3.3
Germany	−1.2	−3.2	−3.7	−3.1	−4.1	−2.8	−3.4
Greece	0.1	−5.1	−3.7	−4.0	−2.8	−4.8	−3.8
Hungary	−0.6	−4.9	−3.7	−6.8	−3.5	−5.2	−4.8
Ireland	−5.1	−4.2	−3.7	−4.3	−8.0	−6.5	−5.6
Italy	0.7	−4.2	−3.7	−3.4	−2.1	−3.4	−3.2
Latvia	−5.9	−3.3	−3.7	−0.8	−8.7	−1.3	−3.6
Lithuania	−2.4	−3.9	−3.7	−4.8	−5.3	−3.8	−4.4
Luxembourg	0.2	−4.2	−3.7	−4.2	−2.6	−11.1	−5.4
Malta	0.6	−4.8	−3.7	−0.6	−2.3	−0.4	−1.8
Netherlands	−1.0	−3.4	−3.7	−3.4	−3.8	−4.1	−3.8
Poland	−3.0	−5.7	−3.7	−7.6	−5.9	−5.8	−5.8
Portugal	1.0	−4.9	−3.7	−3.0	−1.9	−3.0	−2.9
Romania	−1.1	−7.1	−3.7	−8.2	−4.0	−5.5	−5.4
Slovakia	1.8	−4.5	−3.7	−6.1	−1.1	−3.3	−3.5
Slovenia	−1.2	−2.8	−3.7	−3.6	−4.1	−3.0	−3.6
Spain	1.4	−4.5	−3.7	−3.3	−1.5	−3.4	−3.0
Sweden	−2.1	−3.7	−3.7	−0.5	−5.0	−1.1	−2.6
United Kingdom	−2.2	−3.5	−3.7	−1.6	−5.1	−3.1	−3.4

thus filling the actual target in our scenarios. Combination of all countries "easiest" targets, would result in +7% change in emissions compared to 2005 level, and toughest targets, in −22% change, respectively.

When looking at the requirements for improving the greenhouse gas intensity of economy in the non-ETS sector, the relatively fastest improvement is required especially in Luxembourg, Ireland and in some Eastern European countries, Like Poland and Romania (Table 2). However, according to our scenarios, Ireland is the only country that comes close to maintaining the historical rate, on average. Latvia faces great reduction requirements, if emissions are reduced based on reductions in GHG intensity in the past (Scenario 3). Nevertheless, Latvia would be allowed on average less improvement in annual GHG intensity than during 1993–2005. Slovakia, Romania and Poland would face toughest GHG intensity reduction requirements in Scenario 2, equal GHG per GDP. For Sweden, UK, Finland and Denmark, the required effort is less than double the historical rate. For most countries, the EU-proposal rates are close to the average rates in our scenarios.

3.1. *Sensitivities in the results*

Changing assumptions in the scenarios causes a great deal of variation in the reduction targets for countries (Appendix D). Changing the base year from 2005 to 2004 results in mainly minor differences for countries in Scenarios 1 and 3. There is no impact on targets for Scenarios 2 and 4 as these scenarios use projections for 2020 as a basis for calculations (Fig. 2). Countries with lower emissions in the selected base year than their average level are given tougher targets since reductions start from a lower level. Changing the base year from 2005 to 2004 would result in tougher targets for Estonia and Poland, among others, and easier targets for, for example, Slovakia and Portugal.

When the base year for ETS reductions is changed from 2005 to the Kyoto allocations (2008–2012), non-ETS effort increases for all countries by 1–2 percentage points. In 2005 total emissions in the ETS sector were relatively equal to those annually allocated for the 2008–2012 period for most of the countries and at the EU27 level.

Changing the EU's total non-ETS reductions naturally has a great influence on the reduction targets of countries (Fig. 3). If ETS sector emission reductions were smaller, all countries would get tougher non-ETS targets. When ETS reduces 0% instead of −20%, non-ETS sector emissions are allowed to be 85% of the base case non-ETS emissions in 2020 in all scenarios for all countries. However, in terms of additional reduction percentage, variation between countries and scenarios depends on the relation between actual reduction and the size of 2005 emissions. Difference to base case is larger the smaller the relative emission reduction (/bigger the increase) in base case.

Changing the GDP forecast has a great impact on the results in Scenarios 1–3 (an example is SCE1 in Fig. 4). There is no impact on non-ETS reductions when the reductions are based on GHG per capita criteria (SCE4). The reductions with the POLES forecast (Russ et al., 2007) fall closer to the baseline in our study than that of Mantzos et al. (2003). Overall, the forecasts of POLES (Russ et al., 2007) and Mantzos et al. (2003) give tougher targets for eastern European countries compared

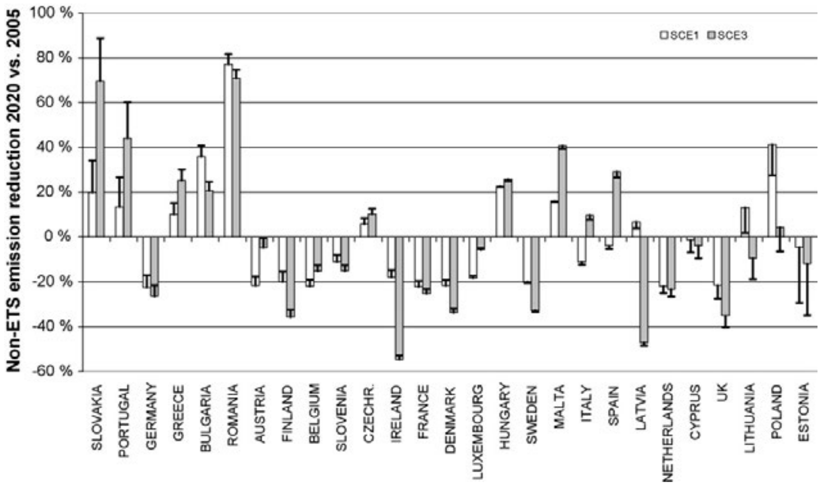


Fig. 2 – Test run 1: Impact of changing the emission base year from 2005 to 2004 in Scenarios 1 and 3. Error bars show the difference in emission reductions compared to the base case scenarios (columns). For Scenarios 2 and 4, there was no impact. For countries furthest left, changing base year from 2005 to 2004 would yield easier targets and countries furthest right, tougher targets.

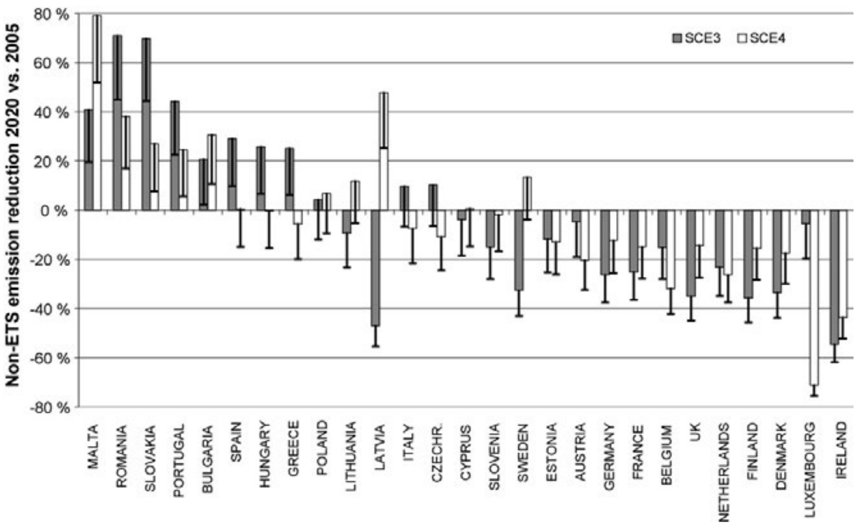


Fig. 3 – Test run 3. Impact of changing the requirement for ETS sector emission reductions from −20% (columns) to 0% (error bars) compared to 2005 level in two example scenarios, 3 and 4. The impact in terms of percentage points is largest on average in these two scenarios for countries furthest left in the figure.

to the base case scenarios with the forecast of Saikku et al. (2008), in which these countries are allowed to grow their emissions. The emission growth until 2020 is thus allowed by the growing economy, even though the GDP/GHG ratio would decrease. Much more moderate rates for these countries in reference forecasts suggest emission reductions between 2005 and 2020. For example, in Saikku et al. (2008), the GDP estimates for 2020 are at least twice as large as in the POLES model for Bulgaria, Latvia and Romania.

The impact of changing the base year for non-ETS GDP between 2004, 2005 or 2008 would be minor, as there is only a 1-percentage point difference for a few Member States and no difference for the others in their share of the overall EU GDP for the different years. Even though the connection between the ETS and non-ETS sectors in terms of emissions and economic growth is somewhat unclear, using overall GDP instead of non-ETS GDP does have a minor impact on the results: there is only a one to two percentage point difference for a few MS and no

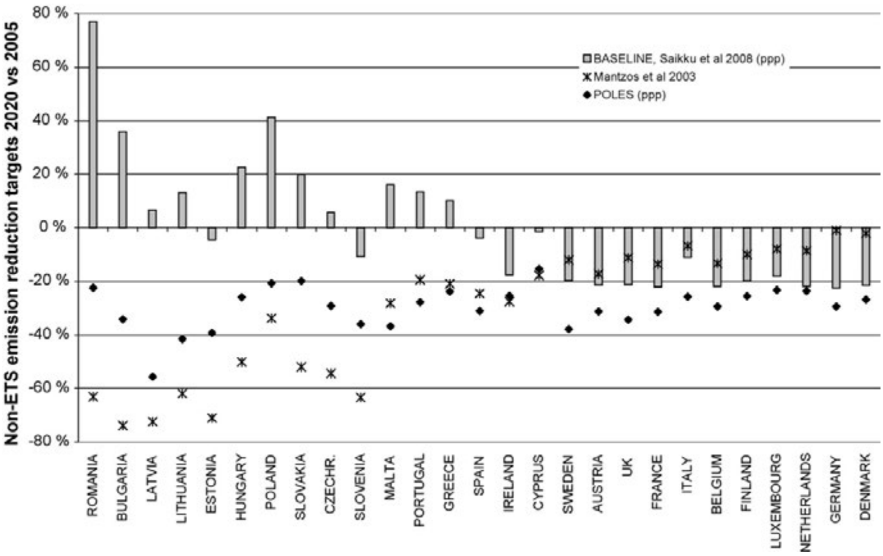


Fig. 4 – Test run 4: Impact of changing GDP forecasts in Scenario 1 (baseline compared to reference forecasts). The reference forecasts have greatest variation compared to the baseline for countries furthest left. SCE1 is shown here as an example, the variation in SCE2 and SCE3 are for most countries the same order of magnitude; for Scenario 4 (equal GHG/capita), the GDP forecast is insignificant.

difference for the others in their individual share of the overall EU GDP.

Changing population forecasts from Eurostat Baseline to Eurostat High or Low did not change the results in Scenarios 1–3 and had only a slight impact on the reduction targets in Scenario 4 (equal GHG per capita in 2020). The High forecast resulted in a 4 or more percentage points difference (lower non-ETS targets in this case) for three countries: Malta, Cyprus and Slovenia. With the Low forecast, Malta and Cyprus gained tougher (more that 4 percentage points) targets.

4. Discussion

To mitigate climate change, the EU has agreed upon a unilateral commitment to reduce GHG emissions by 20% from 1990 levels by 2020. The European Commission has proposed separate targets for ETS and non-ETS sectors, being reductions of 21% and 10%, respectively, compared to the emissions in 2005. The EU Commission has also proposed to share national reduction targets for the non-ETS sector by considering the ability to pay criteria (GDP/capita) and certain extra limitations. For the ETS sector no national quotas are given in the proposal (European Commission 2008).

We studied different ways of sharing the reduction targets among countries and assessed underlying assumptions of the calculations for the period until 2020. Estimates of non-ETS emissions for each EU country for 2020 were generated in alternative ways, which all met the unilateral reduction target as mentioned above for the EU as a whole. We considered unanimous annual reduction, historical development and convergence in GHG/GDP as a basis to share emission targets. In addition, GHG/capita convergence was applied. Different scenarios and changes in underlying assumptions caused great variation in emission reduction targets in this study. The emission reduction requirements for a given country varied depending on the criterion, confirming the findings of den Elzen et al. (2007).

The requirement for the total EU reduction of non-ETS emissions, and hence, the allocation of reduction between ETS and non-ETS, is of great importance. den Elzen et al. (2007) assessed that reducing non-ETS emissions is cost-effective and assumed around 31% reduction from 2005 levels within the non-ETS sector, which is more than the EU proposed. Ekholm et al. (2008) estimated a cost-optimal solution, where nearly equal requirements are set for ETS relative to non-ETS: 14% less in 2020 than in 2005 at both sectors. Much uncertainty is associated to the reductions, as these studies and the EU proposal assume very different emission reduction between the ETS and the non-ETS sectors. Our sensitivity analysis showed that changing the allocation of emission reductions between the ETS and the non-ETS sector has significant influence on the national non-ETS reduction targets of countries, and thus on the reductions and related costs assigned for e.g. agricultural, residential and traffic sectors.

The assumptions behind the calculations are of great importance. The selection of the base year for data used in the calculations may have a significant influence on the results especially if the selected year is very exceptional for a certain country. Using the average values of a prolonged time period rather than one randomly selected year as a starting point for calculations would certainly be more representative.

In addition to statistical sensitivities, more importantly, the choice of GDP forecasts has a major impact on the results. In general, even though forecasts are important when determining emission targets, inequity is embedded in emission allowances when overestimation or underestimation of the future development of GDP occurs. The assumptions behind the Kyoto negotiations compared to the actual development as it took place were inaccurate for some of the Member States such as Finland (Soimakallio et al., 2005). In long-term commitments of emission reduction in nations, in order to mitigate the impact of the uncertainty in forecasts, effort-sharing methods and monitoring mechanisms using some kind of adjustment rules can be considered preparing for unpredictable elements of change.

4.1. *Strengths and weaknesses of the effort-sharing approach studied*

The major strength of simple top-down effort-sharing methods in general is the transparency and limited amount of data required. In addition, statistics for generally known macro-indicators are relatively well-available for different countries. However, at the same time such methods are very limited to take national circumstances explaining the background of macro-figures into account.

GHG emissions, GDP and population were the only statistical data required in the studied approaches in this paper. In addition, we considered forecasts of population and economic growth that can be seen as substantial drivers for the development of emissions. However, also population dynamics, incomes, as well as productive structures and energy intensities of the economy, significantly influence the volume of greenhouse gas emissions, and should be accounted for when allocating emission quotas (Martinez-Zarzoso et al., 2007).

According to Meyerson (1998) population issues were not considered in the formulation of the Kyoto protocol because of the complexity of population interactions as well as political issues. York (2007) explored 14 European nations, finding that population size and age structure have clear effects on energy consumption. Also, economic development and urbanization contribute substantially to changes in energy consumption. Martinez-Zarzoso et al. (2007) found that especially for old EU Member States, the impact of population growth on CO_2 emissions is less than proportional. For New EU countries, however, emissions grow relatively more as population size grows, showing the complexity behind population issues.

Lowering greenhouse gas emissions per economic output or per capita are reasonable targets and inevitably necessary for mitigation of climate change. In principle, a high absolute value for greenhouse gas intensity may depict more inefficient consumption or a more energy intensive structure of economy together with an emission intensive energy production structure. The potential and costs of reducing greenhouse gas intensity may vary extensively between the Member States due to several causes, such as structure of economy, energy production mix, natural resources, climatic and geographical conditions, population density, and public consumption which are not considered in the approaches based on a few macro-figures. Consideration of national features would be important in particular for countries that vary significantly from the average.

Finally, the use of particular macro-figures does not objectively consider consumption. Majority of emission

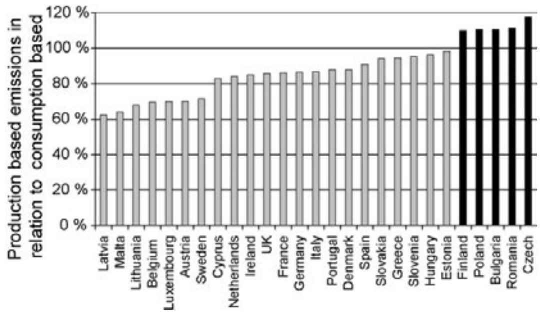


Fig. 5 – Production based GHG emissions (%) in relation to consumption based emissions in 2001 estimated by Peters and Hertwich (2008). Countries with over 100% share (dark columns) are net exporters of embodied emissions in traded goods; others are net importers (grey columns).

inventories allocate emissions to countries based on their production. However, there are significant amounts of emissions embodied in traded goods. According to (Peters and Hertwich, 2008) most countries in the EU27 were net importers of embodied GHG emissions in traded goods (Fig. 5). Only the Czech Republic, Romania, Bulgaria, Poland and Finland were net exporters, i.e. production-based emissions were larger than consumption-based. If consumption based emissions were considered in effort sharing these countries' reduction targets would most likely be lowered. The EU27 as a whole is a net importer; production based emissions covered 89% of the consumption in 2001.

4.2. *Concluding remarks*

The effort all EU members is needed in order to achieve GHG emission reductions of 20% within the European Union by 2020. The required country-specific reductions in the sectors outside emission trade such as transportation, housing, services and agriculture will depend on the applied principle of effort sharing, the allocation of reductions between ETS and non-ETS sectors, the selected base year, and forecasts used. Macro-figures of economy, emissions and population are useful when exploring the trends and targets of future emissions. However, when the aim is to understand the causes for the emissions and reach greater dimensions of equity in effort sharing, a more detailed consideration of the national circumstances may be required to achieve a fair solution. We recommend that different types of indicators and models are used, and assumptions are carefully considered. This provides adequate perspective for the proposed emission reductions. Also, when using forecasts or projections in effort sharing, developing methods that use some kind of rules that allow for adjustment after the primary targets have been set could be valuable for mitigating the impacts of unsuccessful forecasts. Finally, any analysis like this provides relevant information for policy making, but political will, negotiations skills, and the practical capacity of implementing the reductions will eventually determine the success in lowering the GHG emissions as desired.

Acknowledgements

This study was conducted in conjunction with the IFEE (Indicators For Eco-Efficiency, grant 117822) project, which was funded by the Academy of Finland. Funding was also attained from the Finnish Ministry of the Environment. We are grateful to Jaakko Ojala, Magnus Cederlöf, Pekka Kauppi, Ilkka Savolainen and Aapo Rautiainen, and two anonymous reviewers for their comments on this paper.

Appendix A

Total emissions, ETS and non-ETS emissions in EU27 countries 1990 and 2005. In 2020 emissions in the EU27 should be reduced to around 4500 Mt

Member State	Total 1990 emissions (Mt)	ETS 1990 (Mt)	Non-ETS 1990 (Mt)	Total 2005 emissions (Mt)	ETS 2005 (Mt)	Non-ETS 2005 (Mt)	2008–2012 ETS cap allowed (Mt)	Change in total emissions 1990–2005 (%)	Change ETS 1990–2005 (%)	Change non-ETS 1990–2005 (%)
Austria	79.1	37.6	41.5	93.3	33.6	59.7	30.7	18	−11	44
Belgium	145.8	79.8	66.0	143.8	54.6	89.2	58.5	−1	−32	35
Bulgaria	116.1	71.2	45.0	69.8	40.5	29.3	42.3	−40	−43	−35
Cyprus	6	3.4	2.6	9.9	5.0	4.9	5.5	64	48	86
Czech Republic	196.2	125.6	70.6	145.6	83.0	62.6	86.8	−26	−34	−11
Denmark	69	34.0	35.0	63.9	26.2	37.7	24.5	−7	−23	8
Estonia	43.6	33.5	10.1	20.7	12.6	8.1	12.7	−53	−62	−20
Finland	71.2	37.7	33.4	69.3	33.3	36.0	37.6	−3	−12	8
France	564.2	208.2	356.0	553.4	132.8	420.6	132.8	−2	−36	18
Germany	1227.9	695.9	531.9	1001.5	470.7	530.8	453.1	−18	−32	0
Greece	108.7	64.7	44.0	139.2	71.0	68.2	69.1	28	10	55
Hungary	98.7	47.3	51.4	80.5	25.8	54.7	26.9	−18	−46	6
Ireland	55.4	18.8	36.5	69.9	22.4	47.5	22.3	26	19	30
Italy	519.5	263.3	256.1	582.2	227.1	355.1	195.7	12	−14	39
Latvia	26.4	10.7	15.8	10.9	2.8	8.1	3.4	−59	−73	−49
Lithuania	48.1	24.0	24.1	22.6	6.6	16.0	8.9	−53	−73	−33
Luxembourg	12.7	8.1	4.5	12.7	2.5	10.2	2.5	0	−69	124
Malta	2.2	1.5	0.8	3.4	2.0	1.4	2.1	55	35	88
Netherlands	213	109.5	103.5	212.1	80.6	131.5	85.8	0	−26	27
Poland	486.2	313.4	172.9	399	203.5	195.5	208.5	−18	−35	13
Portugal	59.9	29.9	30.0	85.5	36.8	48.7	34.8	43	23	62
Romania	248.7	159.8	88.9	153.7	70.7	83.0	75.9	−38	−56	−7
Slovakia	73	44.4	28.6	48.7	25.3	23.4	30.9	−33	−43	−18
Slovenia	18.4	10.7	7.7	20.3	8.7	11.6	8.3	10	−18	50
Spain	287.4	150.1	137.3	440.6	185.1	255.5	152.3	53	23	86
Sweden	72.2	27.9	44.3	67	19.4	47.6	22.8	−7	−30	7
United Kingdom	771.4	393.5	377.9	657.4	243.2	414.2	246.2	−15	−38	10
EU-27	5620.9	3004.4	2616.6	5177	2122.6	3054.4	2081.0	−8	−29	17

Sources: EC (2007b) for total emissions in 1990, CITL (17 Oct 2007) for ETS/non-ETS sectors' emissions in 2005; EC (2007a) for ETS cap allowed for 2008–2012; Eurostat (2008) for shares of ETS and Non-ETS in 1990, ETS emissions are estimated based on emission from Energy industries, manufacturing industries and construction, and industrial processes.

Appendix B

Population and non-ETS GDP in the EU27 countries, projections for 2008 and 2020 and annual change during 13 years

Country	Population Eurostat (2008)	Population 2020 Eurostat baseline	Annual population growth 2008–2020 (%)	GDP 2008 (M€) Eurostat	GDP 2020 (M€) Saikku et al. (2008)	Annual GDP growth 2008–2020 (%)
Austria	8,211,791	8,441,093	0.2	171,642	220,169	1.9
Belgium	10,504,062	10,790,021	0.2	197,405	252,128	1.9
Bulgaria	7,556,914	6,796,052	−0.8	55,291	122,684	6.3
Cyprus	765,715	865,593	0.9	12,329	19,843	3.7
Czech Republic	10,154,126	9,901,848	−0.2	106,271	183,427	4.3
Denmark	5,446,731	5,526,033	0.1	130,611	167,407	1.9

Appendix B (Continued)

Country	Population Eurostat (2008)	Population 2020 Eurostat baseline	Annual population growth 2008–2020 (%)	GDP 2008 (M€) Eurostat	GDP 2020 (M€) Saikku et al. (2008)	Annual GDP growth 2008–2020 (%)
Estonia	1,327,583	1,247,772	−0.5	18,493	28,832	3.5
Finland	5,269,928	5,404,735	0.2	92,000	120,333	2.1
France	60,985,655	63,571,292	0.3	1,234,587	1,571,287	1.9
Germany	82,753,104	82,676,460	0.0	1,450,534	1,838,186	1.8
Greece	11,199,921	11,427,043	0.2	166,967	300,242	4.6
Hungary	10,028,757	9,693,282	−0.3	90,401	181,062	5.5
Ireland	4,225,110	4,756,111	0.9	111,741	150,108	2.3
Italy	58,532,743	58,299,672	0.0	935,678	1,359,070	2.9
Latvia	2,264,794	2,115,426	−0.5	30,228	52,629	4.4
Lithuania	3,378,964	3,182,215	−0.5	35,067	64,785	4.8
Luxembourg	468,947	520,856	0.8	24,077	32,183	2.3
Malta	415,421	454,020	0.7	5,460	10,348	5.0
Netherlands	16,541,622	17,209,471	0.3	345,072	440,732	1.9
Poland	37,957,353	37,065,252	−0.2	289,189	667,448	6.6
Portugal	10,637,617	10,770,761	0.1	135,369	250,573	4.9
Romania	21,477,014	20,342,159	−0.4	112,641	325,876	8.5
Slovakia	5,359,431	5,270,634	−0.1	42,989	84,210	5.3
Slovenia	2,008,929	2,016,690	0.0	29,740	43,291	2.9
Spain	44,202,506	45,558,613	0.2	682,130	1,070,806	3.5
Sweden	9,116,814	9,575,482	0.4	183,640	240,567	2.1
United Kingdom	60,517,217	62,929,865	0.3	1,383,512	1,777,438	1.9

Sources: Eurostat Database (2008), Saikku et al. (2008).

Appendix C

Non-ETS sector emission reduction targets for 2020 compared to non-ETS emissions in 2005

Principle	GDP/capita EU proposal (%)	Equal annual reduction in non-ETS GHG per GDP SCE1 (%)	Equal non-ETS GHG per GDP in 2020 SCE2 (%)	Historical non-ETS GHG per GDP SCE3 (%)	Equal non-ETS GHG per Capita SCE4 (%)	Average SCE 1–4 Average (%)
Austria	−16	−21	−11	−5	−20	−14
Belgium	−15	−22	−32	−15	−32	−25
Bulgaria	20	36	1	21	31	22
Cyprus	−5	−1	−1	−4	1	−1
Czech Republic	9	6	−29	10	−11	−6
Denmark	−20	−22	7	−34	−17	−16
Estonia	11	−5	−14	−12	−13	−11
Finland	−16	−20	−19	−36	−16	−23
France	−14	−22	−10	−25	−15	−18
Germany	−14	−22	−16	−26	−12	−19
Greece	−4	10	6	25	−6	9
Hungary	10	23	−20	26	0	7
Ireland	−20	−18	−24	−55	−44	−35
Italy	−13	−11	−8	10	−8	−4
Latvia	17	7	58	−47	48	16
Lithuania	15	13	−2	−9	12	3
Luxembourg	−20	−18	−23	−6	−71	−30
Malta	5	16	75	41	79	53
Netherlands	−16	−22	−19	−23	−26	−23
Poland	14	41	−18	4	7	9
Portugal	1	13	24	44	24	27
Romania	19	77	−5	71	38	45
Slovakia	13	20	−13	70	27	26
Slovenia	4	−11	−10	−15	−2	−9
Spain	−10	−4	1	29	0	7
Sweden	−17	−20	22	−33	13	−4
UK	−16	−21	4	−35	−14	−17

Appendix D

Difference in terms of percentage points in test runs compared to the base case. In the test runs, the implications of changing base year for emissions (test run 1), varying ETS allocations (test runs 2 and 3) and changing GDP forecasts (test run 4), changing GDP assumptions (test run 5) and changing population forecasts (test run 6) were studied. Test runs that lead to 2 percentage points difference or less for all countries, are left out from the table (Scenarios 2 and 4 in test run 1, test runs 2 and 5 as a whole, Scenarios 1–3 in test runs 6a and 6b)

	Test run 1		Test run 3				Test run 4 a			Test run 4 b			Test run 6a	Test run 6b
Scenario	1	3	1	2	3	4	1	2	3	1	2	3	4	4
Austria	4	4	−12	−14	−14	−12	4	8	5	−10	8	−12	0	0
Belgium	3	3	−12	−10	−13	−10	8	10	9	−8	9	−8	0	1
Bulgaria	5	4	−21	−15	−19	−20	−110	−81	−97	−70	−40	−62	1	−1
Cyprus	−5	−6	−15	−15	−15	−15	−16	−14	−16	−14	7	−14	5	−5
Czech Republic	3	2	−16	−11	−17	−14	−60	−39	−63	−35	−12	−36	1	0
Denmark	2	2	−12	−16	−10	−13	19	31	16	−5	18	−4	−1	0
Estonia	−25	−23	−15	−13	−14	−13	−66	−59	−61	−35	−18	−32	3	−3
Finland	5	3	−12	−12	−10	−13	10	13	8	−6	13	−4	−1	1
France	3	2	−12	−14	−11	−13	8	13	8	−9	9	−9	−1	1
Germany	6	5	−12	−13	−11	−13	21	26	20	−7	11	−7	0	−1
Greece	5	5	−17	−16	−19	−14	−31	−28	−35	−34	−15	−38	−1	−1
Hungary	0	−1	−19	−12	−19	−15	−73	−46	−75	−48	−20	−50	1	−1
Ireland	3	2	−13	−12	−7	−9	−10	−7	−5	−8	10	−4	0	0
Italy	−1	−2	−14	−14	−16	−14	4	7	5	−15	4	−18	0	1
Latvia	−3	−1	−16	−24	−8	−22	−79	−116	−39	−62	−76	−31	3	−2
Lithuania	−11	−9	−17	−15	−14	−17	−75	−64	−60	−55	−35	−44	2	−2
Luxembourg	1	1	−12	−12	−14	−4	10	12	12	−5	13	−6	0	−1
Malta	0	−1	−18	−27	−21	−27	−44	−63	−54	−53	−56	−64	7	−6
Netherlands	−3	−3	−12	−12	−12	−11	13	17	13	−2	18	−2	0	0
Poland	−14	−11	−21	−13	−16	−16	−75	−43	−55	−62	−25	−46	1	−1
Portugal	13	16	−17	−19	−22	−19	−33	−33	−42	−41	−25	−52	0	−1
Romania	5	4	−27	−14	−26	−21	−140	−74	−135	−100	−43	−96	0	−1
Slovakia	14	19	−18	−13	−25	−19	−72	−51	−102	−40	−15	−56	0	−1
Slovenia	3	3	−14	−14	−13	−15	−53	−52	−50	−25	−9	−24	4	−3
Spain	−1	−3	−15	−15	−19	−15	−21	−19	−28	−27	−11	−36	0	0
Sweden	−1	−1	−12	−19	−10	−17	8	16	7	−18	−4	−15	−1	0
UK	−6	−5	−12	−16	−10	−13	10	17	8	−13	4	−11	0	0

REFERENCES

Blok, K., Phylipsen, G.J.M., Bode, J.W., 1997. The triptique approach. Burden differentiation of CO_2 emission reduction among European Union member states. Discussion Paper for the Informal Workshop for the European Union Ad Hoc Group on Climate Zeist, The Netherlands, 16–17 January, 1997.

CITL, 2007. Community Independent Transaction Log Available at: http://ec.europa.eu/environment/climat/emission/citl_en.htm.

Delhotal, K.C., Chesnaye, F.C.de la, Gardiner, A., Bates, J., 2006. Mitigation of methane and nitrous oxide emissions from waste, energy and industry. The Energy Journal 3, 45–62 (special issue).

den Elzen M.G.J., Lucas P.L., Gijsen A., 2007. Exploring European countries' emission reduction targets, abatement costs and measures needed under the 2007 EU reduction objectives. MNP Report 500114009/2007. Netherlands Environmental Assessment Agency.

Ekholm, T., Lehtilä, A., Savolainen, I. 2008. Unilateral emission reductions of the EU and multilateral emission reductions of the developed countries—assessing the impact on Finland with TIMES model (in Finnish with English abstract). VTT Working Papers 96. Espoo, Finland, March 2008, 57 p.

European Commission, 2007a. National Allocation Plans: Second Phase (2008–2012). Available at: http://ec.europa.eu/environment/climat/2nd_phase_ep.htm.

European Commission, 2007b. Commission staff working document. Accompanying document to the communication from the commission. Progress towards achieving the Kyoto objectives. COM(2007)757 Final. SEC(2007) 1576, Brussels, 27.11.2007.

European Commission, 2008. Package of Implementation Measures for the EU's Objectives on Climate Change and Renewable Energy for 2020, Brussels.

Eurostat, 2008. Eurostat Database. Available at: ec.europa.eu/eurostat.

Heston, A., Summers, R., Aten, B., 2007. Penn World Table, Version 6.2. Center for International Comparisons of Production, Income and Prices at the University of Pennsylvania. http://pwt.econ.upenn.edu/.

IEA, 2006. CO_2 emissions from fuel combustion 1971–2004. International Energy Agency, OECD/IEA, 2006.

IPCC, 2007. Climate Change 2007. The Physical Science Basis. Summary for Policymakers. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC secretariat Switzerland.

Mantzos, L., Capros, P., Kouvaritakis, N., Zeka-Paschou, M., 2003. European Energy and Transport Trends to 2030, EU DG-TREN, Brussels, Belgium.

Martinez-Zarzoso, I., Bengochea-Morancho, A., Morales-Lage, R., 2007. The impact of population on CO_2 emissions: evidence from European countries. Environmental & Resource Economics 38 (4), 497–512.

Meyerson, F.A.B., 1998. Population, carbon emissions, and global warming: the forgotten relationship at Kyoto. Population and Development Review 24, 115–130.

Peters, G., Hertwich, E., 2008. CO_2 embodied in international trade with implications for global climate policy. Environmental Science and Technology 42 (5), 1401–1407.

Phylipsen, G.J.M., Bode, J.W., Blok, K., Merkus, H., Metz, B., 1998. A Triptych sectoral approach to burden differentiation; GHG emissions in the European bubble. Energy Policy 26 (12), 929–943.

Phylipsen, D., Höhne, N., Janzic, R. 2004. Implementing Triptych 6.0—Technical Report. DM 70046/ICC03080. Commissioned by RIVM, November 2004, 56 p.

Russ, P., Wiesenthal, T., van Regemorter, D., Ciscar, J., 2007. Global Climate Policy Scenarios for 2030 and beyond Analysis of Greenhouse Gas Emission Reduction Pathway Scenarios with the POLES and GEM-E3 Models. European Commission Joint Research Centre, Institute for Prospective Technological Studies, Sevilla, Spain.

Saikku, L., Rautiainen, A., Kauppi, P.E., 2008. The sustainability challenge of meeting carbon dioxide targets in Europe by 2020. Energy Policy 36 (2), 730–742.

Sijm, J.P.M., Berk, M., den Elzen, M., van den Wijngaart, R., 2007. Options for Post-2012 EU Burden Sharing and EU ETS Allocation Climate Change Scientific Assessment and Policy Assessment. ECN & Netherlands Environmental Agency, Netherlands.

Soimakallio, S., Savolainen, I., Syri, S., 2005. GHG emission development in the EU and assessment of the Triptych approach applicability for the EU internal burden sharing. Extended English Summary. VTT Project Report no. PRO3/P54/04, Helsinki, Finland, 32 p.

Soimakallio, S., Perrels, A., Honkatukia, J., Moltmann, S., Höhne, N. 2006. Analysis and Evaluation of the Triptych 6—Case Finland. VTT Working Papers 48, Espoo, 2006. http://www.vtt.fi/inf/pdf/workingpapers/2006/W48.pdf.

Torvanger, A., Ringius, L., 2001. Burden Differentiation: Criteria for Evaluation and Development of Burden Sharing Rules. CICERO Center for International Climate and Environmental Research, Oslo, Norway.

York, R., 2007. Demographic trends and energy consumption in European Union Nations, 1960–2025. Social science research 36 (3), 855–872.

Laura Saikku is a PhD student in environmental science and policy at the University of Helsinki. She holds a Master's degree in agriculture and forestry. Her research interests are in the field of industrial ecology. Her recent research focuses on European climate policy and the challenge of meeting emissions reduction targets, as well as on the nutrient flows of the human economy.

Sampo Soimakallio, Master of science in technology, is working as a senior research scientist at the VTT Technical Research Centre of Finland. He has been involved in analyzing various principles in the differentiation of emission reduction commitments among countries. He is also working with climatic sustainability of bio-fuels.

PAPER VI

Effort sharing in ambitious, global climate change mitigation scenarios

In: Energy Policy 38(4), 1797–1810.

Energy Policy 38 (2010) 1797–1810

Contents lists available at ScienceDirect

Energy Policy

journal homepage: www.elsevier.com/locate/enpol

Effort sharing in ambitious, global climate change mitigation scenarios

Tommi Ekholm [a,c,*], Sampo Soimakallio [a], Sara Moltmann [b], Niklas Höhne [b], Sanna Syri [a], Ilkka Savolainen [a]

[a] VTT Technical Research Centre of Finland, P.O. Box 1000, FIN-02044 VTT, Finland
[b] Ecofys Germany GmbH, Cologne, Germany
[c] TKK Helsinki University of Technology, Espoo, Finland

ARTICLE INFO

Article history:
Received 26 June 2009
Accepted 19 November 2009
Available online 23 December 2009

Keywords:
Climate change mitigation
Effort sharing
Scenario

ABSTRACT

The post-2012 climate policy framework needs a global commitment to deep greenhouse gas emission cuts. This paper analyzes reaching ambitious emission targets up to 2050, either -10% or -50% from 1990 levels, and how the economic burden from mitigation efforts could be equitably shared between countries. The scenarios indicate a large low-cost mitigation potential in electricity and industry, while reaching low emission levels in international transportation and agricultural emissions might prove difficult. The two effort sharing approaches, Triptych and Multistage, were compared in terms of equitability and coherence. Both approaches produced an equitable cost distribution between countries, with least developed countries having negative or low costs and more developed countries having higher costs. There is, however, no definitive solution on how the costs should be balanced equitably between countries. Triptych seems to be yet more coherent than other approaches, as it can better accommodate national circumstances. Last, challenges and possible hindrances to effective mitigation and equitable effort sharing are presented. The findings underline the significance of assumptions behind effort sharing on mitigation potentials and current emissions, the challenge of sharing the effort with uncertain future allowance prices and how inefficient markets might undermine the efficiency of a cap-and-trade system.

1. Introduction

The ambitious climate change mitigation targets considered currently require global participation in the mitigation effort in the post 2012-period. Article 3.1 of the United Nations Framework Convention on Climate Change (UNFCCC) requires that the mitigation effort should be shared between the parties "on the basis of equity and in accordance with their common but differentiated responsibilities and respective capabilities". In order to reach a global solution, the equity issue has to be solved. Each country has to have the impression that it is treated equitably relative to the others in order for it to participate.

The question of what is actually equitable is ambiguous, and Article 3.1 is thus open to interpretations. As an example, Ringius et al. (1998) lists the following equity concepts:

- Egalitarian—equal emissions per capita.
- Sovereign—equal reductions from, e.g., 2000.
- Horizontal—equal net change in welfare, e.g. in GDP.
- Vertical—effort dependent on ability.
- Equal responsibility—effort based on historical emissions.

In addition to equity, to achieve economic efficiency the emissions should be mitigated where least costly. Solutions to the conflict between equity and efficiency include cap-and-trade systems or harmonized emission taxes. Under perfect markets without uncertainty, the approaches should produce the same outcome. The equity issue can then be dealt with either the allocation of tradable emission allowances or the redirection of tax revenues. Due to a more simpler setting, this paper analyzes a global cap-and-trade system.

In a perfect market setting the allocation of emission allowances is merely a financial compensation. The parties are free to trade allowances and their actions are guided solely by the market price of allowances, not by how much the party initially owns allowances. Therefore in principle the mitigation costs of the parties could be adjusted through the allocation without affecting the actual mitigation measures.

The level to which the global emissions should be reduced is obviously debatable. However, as were shown by Manne and Stephan (2005), under certain conditions, the optimal level of abatement for different countries does not depend on the

* Corresponding author at: VTT Technical Research Centre of Finland, P.O. Box 1000, FIN-02044 VTT, Finland. Tel.: +358 40 775 4079; fax: +358 20 722 7026.
E-mail address: tommi.ekholm@vtt.fi (T. Ekholm).

0301-4215/$ - see front matter © 2009 Elsevier Ltd. All rights reserved.
doi:10.1016/j.enpol.2009.11.055

allocation of allowances. Therefore the overall abatement level and equity issues can be separated and analyzed on their own.

Given an overall emission limit, effort sharing deals with the distribution of limited emission allocations to the parties. The effort sharing process and tools used should be reliable, understandable and transparent in order to build confidence in the process. The resulting allocations, however can, and moreover should, be analyzed with more sophisticated if less transparent models.

This paper focuses on the equity of effort sharing with two exogenously assumed reduction targets that would stabilize greenhouse gas atmospheric concentrations to 485 ppm CO_2-eq and 550 ppm CO_2-eq by the end of the century. A simple and transparent tool Evolution of Commitments (EVOC) (Höhne et al., 2006) tool is used to calculate the allocation of emissions, which are then used in long-term energy-climate scenarios produced with ETSAP-TIAM (Loulou and Labriet, 2008; Loulou, 2008), a more sophisticated integrated assessment model. Though transparently documented, the TIAM may be seemingly opaque due to its size and complexity.

The stance of vertical equity with respect to economic burden from mitigation is taken here, reflecting the ‘‘respective capabilities’’ stated in Article 3.1. Then the effort sharing rule should allocate higher mitigation costs (relative to GDP) for wealthier countries, measured e.g. in terms of GDP per capita, much in the same sense as progressive taxation taxes more those with higher income. The mitigation costs considered include direct mitigation costs, changes in energy trade, allowance trade and the value of lost demand due to price elasticity; but disregard indirect macroeconomic costs, damage costs and possible benefits from avoided climate change.

Numerous mitigation scenario studies have already been made. Past studies have, however, often considered only CO_2 or higher stabilization levels for atmospheric greenhouse gas concentrations than what can currently be seen as relevant (Fisher et al., 2007). Also, a number of studies investigating the effort sharing have been conducted. The studies have, however, analyzed the effort sharing only in terms of allocated emissions and by comparing them to GDP, historical emissions or population (Miketa and Schrattenholzer, 2006; Vaillancourt and Waaub, 2004), taken only CO_2 into account (Persson et al., 2006; Russ et al., 2005), or used a simplified model with marginal abatement curves (MACs), as e.g. den Elzen et al. (2005, 2007) with the FAIR model. An exception from these, though, is den Elzen et al. (2008b), which evaluates two effort sharing rules with the FAIR model using updated MAC curves, and including also detailed analyses with the energy and land use models of IMAGE. Studies with general equilibrium models have also been carried out (Böhringer and Welsch, 2004; Peterson and Klepper, 2007) providing light on the macroeconomic effects of mitigation measures, though with less detail on specific mitigation measures.

This paper intends to address these shortcomings with a threefold purpose. First, the attainability of ambitious mitigation targets, -50% from 1990 levels, for all Kyoto-gases until 2050 are analyzed while also exploring possible bottlenecks for further mitigation. Second, the mitigation scenarios are used to evaluate two effort sharing rules, also extending the analysis of effort sharing from past studies with regard to the equitability issue. Given the varying sectoral distribution of emissions across countries, the explicit reporting of the mitigation measures in the scenarios is also significant for effort sharing. Third, challenges in effort sharing are also analyzed, including imperfect allowance markets and consideration of uncertainties.

The paper is structured as follows. Section 2 describes the models for producing the emission allocations and the energy-climate scenarios along with some main assumptions. Section 3 first outlines the main mitigation measures in the scenarios, then focuses on the main economic outcomes both in global and regional scale, and finally assesses the equity of effort sharing. In Section 4 the relevant uncertainties, two cases of allowance market imperfections and the importance of assumptions behind the effort sharing are considered. Last, Section 5 draws up conclusions and discusses the main findings.

2. The models and scenario assumptions

Two separate models were used in this study. First, EVOC, a transparent but simplified effort sharing tool of Ecofys GmbH, is used to quantify the emission allocation with the Triptych (Phylipsen et al., 1998) and Multistage (den Elzen et al., 2006) effort sharing regimes. Future energy-climate scenarios with the two reduction targets are then analyzed with the more sophisticated but complex ETSAP-TIAM, a global integrated assessment model of the TIMES family. Although the TIAM is well documented, fully consistent and the input data can be made available upon request, the vast size and relative complexity of the model may render the model non-transparent to the reader.

2.1. EVOC

The effort sharing is based on Triptych and Multistage calculations from EVOC (Höhne et al., 2006). These effort sharing approaches were chosen as subjects as the Triptych approach might provide a good balance between simplicity and detail, and Multistage might provide a relevant ’’ladder’’ for developing countries to join. EVOC contains collections of data on emissions from several sources and future projections of relevant variables from the IMAGE implementation of the IPCC SRES scenarios. As emission data vary in its completeness and sectoral split, EVOC combines data from the selected sources and harmonizes it with respect to the sectoral split.

Future emissions are based on IMAGE projections of parameters, such as population, GDP (PPP), electricity consumption and industrial value added. As IMAGE projections are available only for 17 world regions, EVOC de-aggregates these data by combining it with historical values. Finally, the user can set the parameters of several effort sharing rules in order to calculate emission allocations. The main parameters used in this study are provided in the electronic annex for the paper in the publisher’s website.

2.1.1. Triptych

The Triptych approach was originally developed for sharing the CO_2 mitigation effort between the EU member states using three sectors: power sector, the internationally operating energy-intensive industry and the domestically oriented sectors (Phylipsen et al., 1998), but has been updated thereafter to contain more countries (Groenenberg et al., 2001), sectors and greenhouse gases, and recently also to have multistaged commitments (den Elzen et al., 2008a).

The emission target for each sector is calculated with given assumptions on the reduction potentials in the sector. The Triptych version 6.0 that was used in the study is documented by Phylipsen et al. (2004). This version uses six sectors: Electricity, Industry, Fossil fuel production, Domestic, Agriculture and Waste. The electricity and industry sectors use parameters on efficiency, structure and income levels to calculate the emission limits. Domestic, and waste sectors use a single convergence level, given in terms of $tCO_2-eq/capita$, to which the emissions of countries converge by a given year. This is to reflect the converging living standards and practices in different countries. For fossil fuel production and agriculture, reduction levels from the baseline are assumed. In addition to this sectoral differentiation, Triptych also uses a rough income categorization with some parameters to distinguish countries with different levels of affluence.

The emission allocation of a country is then the sum of the sectoral targets. It is though critical to note that only the country level target is binding, not the sectoral targets on which the country level target is based on. Thus Triptych is not a sectoral approach per se, but uses sectoral mitigation potentials to arrive on a more accurate estimate on how much reductions are feasibly attainable in a given country and leaves the country free to choose how to pursue its target. As the Triptych approach takes into account the sectoral distribution of emissions, and even though it uses in principle uniform sectoral potentials across all countries, it has the ability to accommodate national circumstances better than most other simplified approaches. It also explicitly allows for economic growth and improving efficiency in all countries and aims to put internationally competitive industries on the same level.

2.1.2. Multistage

As the name suggests, in a Multistage approach the countries participate in several stages with differentiated levels of commitment (den Elzen et al., 2006). Each stage has stage-specific commitments with countries graduating to higher stages when they exceed certain thresholds (e.g. emissions per capita or GDP per capita), and all countries agree to have commitments at a later point in time. For this study, thresholds and commitments based on per capita emissions with four stages were applied.

Least developed countries start at stage 1, which carries no commitments. At stage 2 the countries commit to sustainable development, in practice moderate reductions, e.g. 10%, from the baseline scenario. Stage 3 would involve moderate absolute targets, e.g. more stringent targets than in stage 2. The target could now also be only positively binding, so that the country could sell allowances if it reaches its target but would not be penalized if it did not. Finally, at stage 4 the country faces substantial reduction targets. As time progresses, more and more countries enter the stage 4.

In this study, the concept of Multistage effort sharing is, however, slightly abused, as the cap-and-trade system was assumed to bind all countries. Instead, the countries without binding commitments receive emission allocations according to their baseline emissions, but are then free to mitigate emissions and sell the excess allowances for profit. If this were not the case, the mitigation policy regime would lose its effectiveness.

2.2. ETSAP-TIAM

The energy and emission scenarios in the study were formed with the TIAM (TIMES Integrated Assessment Model) (Loulou and Labriet, 2008; Loulou, 2008), which is based on the TIMES (The Integrated MARKAL-EFOM System) modelling methodology (Loulou et al., 2005a), both developed under the IEA's Energy Technology Systems Analysis Program (ETSAP). The TIMES family of models are bottom-up type linear partial equilibrium models that calculate the market equilibrium through the maximization of the total discounted economic surplus with given external end-use demand projections. The models assume perfect markets and, in their basic form, unlimited foresight for the calculation period.

The TIAM models the whole global energy system with 15 geographical regions. Main assumptions concerning the energy system, future energy technologies, potentials and other mitigation options in the model are described by Syri et al. (2008). All Kyoto-greenhouse gases (CO_2, CH_4, N_2O and F-gases) from all anthropogenic sources are covered by the model, although emissions from land use change were not considered in this study.

The energy consumption is based on external projections of the growth of regional GDP, the population and the volume of various economic sectors, which have been harmonized to the IMAGE implementation of four SRES scenarios that are used in EVOC, ensuring consistency between the models. Inclusion of four different energy demand scenarios—marked as A1, A2, B1 and B2—provides also perspective on the effect of different assumptions on energy demand in the future.

In order to satisfy the demands, the model contains estimates on energy resources, a vast number of technology descriptions for energy production, transformation and end use, and a number of other elements, such as user-defined constraints. The flows and prices of energy commodities, including international trade for energy and emission allowances, are calculated endogenously by the model.

The model also uses price-elasticity for energy end-use demand in the mitigation scenarios, so that final energy demand reacts to changing energy prices compared to the baseline scenario. The demand elasticity for changes in energy prices was assumed to be moderate, around -0.2 for most demands and around -0.4 for aviation and maritime transport, which were assumed to be more affected by changes in energy use prices. These values are very similar to the values used by e.g. Loulou et al. (2005b) or Persson et al. (2006) with similar models. Due to this elasticity, the model can take macroeconomic feedbacks into account in a limited manner, and allows the model to reach the emission targets with lower costs than with inelastic demand. A sensitivity analysis on this by Persson et al. (2006) indeed confirmed this, and suggested that there might be also considerable regional variation in the effect of elasticity on mitigation costs. Therefore further work on the issue might be appropriate.

The model also includes a simplified climate module (Syri et al., 2008; Loulou and Labriet, 2008) that calculates changes in radiative forcing and global mean temperature with the resulting emissions. The module uses three reservoirs for CO_2 in the biosphere, first-order decay models for CH_4 and N_2O, and two heat reservoirs for calculating the temperature change. F-gases are converted into CO_2 equivalents while calculating the concentrations.

2.3. Main scenario assumptions

In addition to the technological and resource assumptions made in the TIAM model, assumptions on socio-economic development and the effort sharing itself are obviously important. As has many times been previously noted, e.g. in Riahi et al. (2007), the abatement effort is very dependent on the baseline scenario. With higher energy demand and emission projections, it is harder and costlier to meet a stringent emission target. Four different economic and population growth projections from the IMAGE implementation of the SRES scenarios (IPCC, 2000) were used consistently in both EVOC and TIAM. The growth of global GDP varies in the scenarios from 2.3% to 3.6% p.a. between 2000 and 2050 with regional growth rates being higher for developing and lower for developed countries. The projections were used to project the end-use energy demand in the baseline scenarios, to which the mitigation scenarios were compared to in order to calculate the mitigation costs.

Main characteristics of the two reduction targets considered are presented in Table 1. The targets were assumed to be globally binding from 2020. For calculating the resulting concentrations, radiative forcing and mean temperature increase (using 3 °C

Table 1
The implications of the two emission targets used.

Concentration in 2100	485 ppm	550 ppm
Emissions from 1990 in 2020	+20%	+30%
Emissions in 2020 (Gt CO_2−eq)	37.1	39.5
Emissions from 1990 in 2050	−50%	−10%
Emissions in 2050 (Gt CO_2−eq)	15.4	28.2
Rad. forcing in 2100 (W/m^2)	3.0	3.6
Temp. increase in 2100 (°C)	1.8	2.1

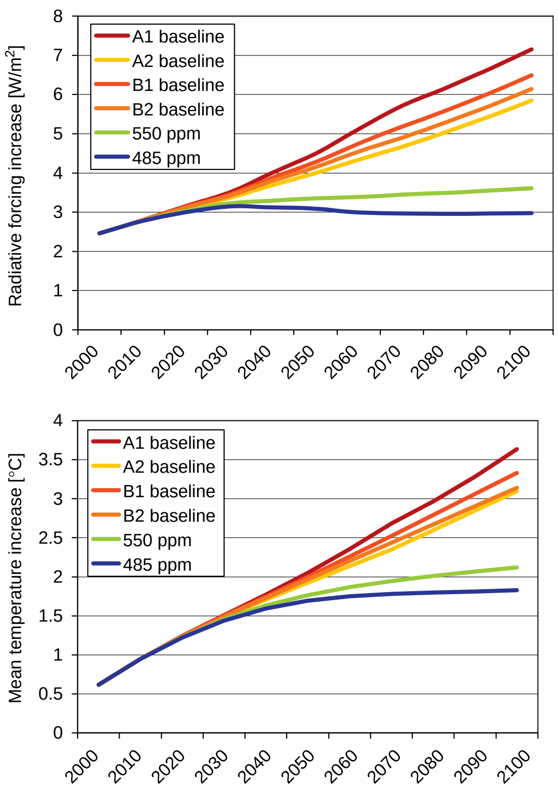


Fig. 1. Increase in radiative forcing (W/m^2, top) and global mean temperature (°C, bottom) in the four baseline scenarios and with the two mitigation scenarios.

climate sensitivity) up to 2100, the emission target of 2050 was assumed constant for the period between 2050 and 2100. If further reductions would be made post-2050, though, concentrations below 485 and 550 ppm would be attainable by 2100.

The more stringent target falls in the high end of IPCC Category I of stabilization levels (Fisher et al., 2007). It overshoots first to 505 ppm CO_2−eq in 2030 before declining to levels around and below 490 ppm, as can be seen in Fig. 1. The figure also presents the global mean temperature increase in baseline and reduction cases. With the 485 ppm target the temperature stabilizes during the century, whereas with the 550 ppm target it is still increasing in 2100 and would probably stabilize around 2.5 °C later on.

It is, however, critical to note that the measures in the scenarios do not affect land use change and forestry emissions. An undisturbed baseline scenario was assumed for deforestation, thus increasing the overall CO_2 emissions and concentrations. As the focus here is on effort sharing, and as the uncertainties of both deforestation emissions and afforestation measures are very large, it was natural to disregard these.

Fig. 2 presents the emission allocation, relative to 2000 emissions, in 2020 and 2050 for the 15 different countries or country groups in TIAM. The bars present the median of the four economic growth scenarios. The approaches allocate, respectively, 10–50% reductions for Annex I in 2020 and 60–95% reductions in 2050. Non-Annex I regions may increase their emissions up to 2020 by varying amounts, whereas in 2050 only the least developed regions receive allocations above their 2000 emission levels. Also it can be noted that the Multistage approach generally allocates more emissions to the least developed countries in 2050 than Triptych.

3. Scenarios

3.1. Emissions and mitigation measures

Of all the eight different mitigation scenarios created, the moderate growth B2 scenarios with both reduction targets are used for illustrating the mitigation measures. Fig. 3 portrays the emission profiles in both cases, separately for combustion and process emissions. As can be seen from Fig. 3, the electricity sector provides the largest cost-efficient mitigation potential. Also large emission reductions are carried out in the industrial sector and a number of measures also in the other sectors. Below is a list of main measures in five sectors:

- Electricity: Phase-out of coal; strong adoption of wind power and biomass; slight increase in hydro and nuclear from baseline; gas and coal with CCS.
- Industry: Phase-out of fossil fuels, especially coal; CCS; biomass, also combined with CCS; N_2O from chemical industries; blended cements replacing clinker.
- Transportation: Fuel efficiency; natural gas on heavier road vehicles; later hydrogen or electricity.
- Residential and commercial: The energy mix shifting to electricity and heat; efficiency; considerable potential on waste CH_4.
- Agriculture: Limited low-cost potential in all categories; extensive reductions challenging e.g. in cattle and rice paddy CH_4 and soil N_2O.

3.1.1. Electricity and industry

Emission reductions in electricity production and industry are perhaps the most straightforward and extensively studied. Phase-out of coal and other fossil fuels, or their use in conjunction with CCS, would contribute to the most of the emission reductions. Also, sustainably grown bio-energy with CCS could provide negative emissions.

Most electricity generating technologies, such as wind power, nuclear energy and biomass, are mature and already in the market. In the medium-long term, the only technology currently still in the demonstration phase is CCS. In 2050, however, there would be a need for novel production technologies as fusion power, though being very costly, emerged in 2050 in the scenarios, especially with the 485 ppm target.

Changes and improvements in industrial processes, such as increased use of steel scrap or inert anodes in aluminium smelters, would also contribute to the reductions. Blended cement and clinker kilns with CCS could be used in cement production. Also, N_2O emission reductions using thermal destruction and catalytic

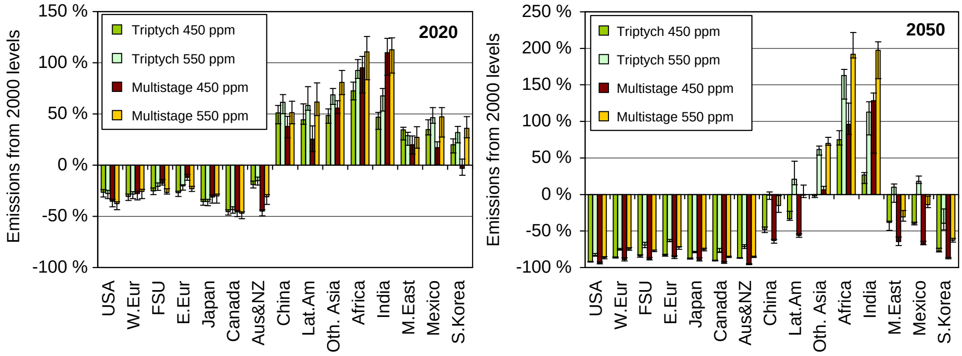


Fig. 2. Emission allocation, relative to 2000 emissions, with the Triptych and Multistage effort sharing approaches and two reduction targets in 2020 (left) and 2050 (right). The error bars correspond to the range of values with four baseline scenarios.

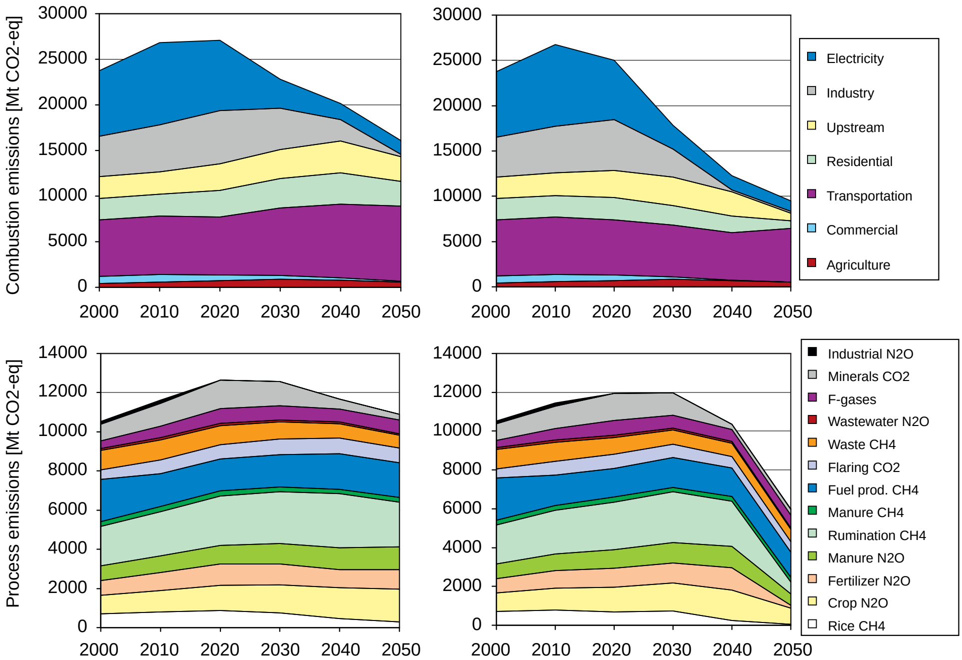


Fig. 3. Global greenhouse gas emissions with the 550 ppm (left) and 485 ppm (right) mitigation targets, split between combustion (top) and process-based (bottom).

reduction, respectively, in adipic and nitric acid industries are one of the first mitigation measures taken.

The total energy consumption in industry is reduced by roughly 8% in 2020 compared to the baseline due to better energy efficiency, leaving total industrial output down 2–3% from the baseline due to the demand price-elasticity. The rising carbon price affects production in the long run, and industrial production is on average 12% below the baselines with the 485 ppm target in 2050. With a 2% annual growth rate in industry output, this would equal a rather small 0.25% reduction in the annual growth rate.

3.1.2. *Transportation*

In road transportation deep reduction through a shift to natural gas, electricity/hydrogen and biofuels (when sustainably produced) should be feasible. Rising demand could however turn the decreasing trend in road transportation emission again to a rise by 2050 even with the low-emission technologies.

International transportation—especially aviation—might also pose more difficulties. Even though the fuel efficiency has improved in aviation, development extrapolated from the historical pace is not sufficient to stabilize emissions with the projected growth in aviation demand (Macintosh and Wallace, 2009). Clearly, then, if the emissions are deemed to decrease, also the demand has to decrease to some extent.

Studies and demonstrations with liquid hydrogen and biofuels (Fischer–Tropsch kerosene) as alternative aviation fuels have been conducted. Both fuels, however, have their difficulties. Hydrogen airplanes involve large technical and operational challenges due to the low volumetric energy density and the need for pressurized cryogenic tanks. The Fischer–Tropsch process is technologically mature and the product resembles fossil kerosene. The challenge with biofuels is, however, of price and quantity. The baseline final energy demand for aviation and shipping equalled roughly 60 EJ/a in 2050. As the required primary energy would be higher, it might prove hard to increase sufficiently the bioenergy supply—roughly at 130 EJ/a in the 485 ppm scenarios in 2050—even though the rising allowance prices might render biofuels competitive.

Due to these challenges, the technologies were excluded from the scenarios, and, as a result, the level of transportation emission remains relatively constant throughout the 485 ppm scenario.

3.1.3. *Agriculture*

Important mitigation potential exists in agriculture, often in the form of improved management practices. Mitigation measures have been analyzed for example in the EMF-21 study (DeAngelo et al., 2006), on which the mitigation measures in the TIAM model are mostly based on. The applicability of most measures is, however, only partial, and agricultural emissions tend to continue their growth in the reduction scenarios.

When very stringent emission targets, such as -50% reductions from 1990, are pursued, also agricultural emission have to be reduced considerably. If the potentials of technological and management options do not improve substantially from those assessed in DeAngelo et al. (2006), a shift towards less emitting agricultural products, e.g. cattle to poultry and swine and rice to other cereals, might be necessary.

With sufficiently high allowance prices this might happen directly through the market mechanism. As an example, assuming emissions of 1.5 t CO_2–eq/head/a (IPCC, 2006) for beef cattle and 200 kg meat yield after two years, an allowance price of 500$\$_{2000}$/t CO_2 would increase the producer price by 7.5$\$_{2000}$/kg meat. Similarly, taken the default emission factor of 1.3 kg CH_4 /ha/d for rice paddy (IPCC, 2006) and a production of 4 t rice/ha/a (FAO, 2009), the producer price would increase by 1.2$\$_{2000}$ /kg rice due to the emissions. Being roughly 2–5 and 10 times higher than the producer prices in 2000 (FAO, 2009), respectively, for cattle meat and rice, price increases of this magnitude might cut consumption considerably and shift it to lower emitting substitutes.

As the emission sources are very dispersed and mostly concentrated on rural areas of less developed countries, it is harder to control the emissions and effectively introduce better practices. Also, it is important to note the major uncertainties and dependences on local conditions with agricultural emissions, especially concerning N_2O.

A very important source of potential mitigation measures, reduced deforestation and afforestation, were not considered in the scenarios. As the estimates both on emissions from deforestation and mitigation options are very uncertain, these emissions and mitigation measures might distort the analysis of effort sharing substantially. On the global scale, the exclusion of these measures, however, increases the mitigation costs in the scenarios, perhaps even drastically.

3.2. *Mitigation costs*

The main issue in effort sharing is how to divide the global mitigation costs between the countries. Clearly, an important factor here is the total level of costs. The effect of different baseline scenarios and reduction targets on the mitigation costs has been noted in previous studies (e.g. Riahi et al., 2007). This arises from different demand levels for end-use commodities and the system costs in the baseline scenario.

An often used measure of economic burden is the mitigation costs, i.e. the difference in energy system costs between baseline and mitigation scenarios, divided by the projected global GDP. Fig. 4 portrays this measure on global scale in 2020 and 2050 for a spectrum of mitigation targets and four socioeconomical scenarios. The more ambitious end of the reduction targets equals the 485 ppm mitigation target and the more lax the 550 ppm target, the targets between being linear interpolations of the 485 and 550 ppm targets.

As the economic burden of mitigation is shared through the allocation and trade of emission allowances, the price of allowances is critical for effort sharing. Fig. 5 portrays the average price of allowances between 2020 and 2050 in the scenarios with both mitigation targets. As can be seen from the figure, the price is projected to rise steeply after 2030 with the tightening emission limits, especially with the 485 ppm target.

3.3. *Effort sharing*

Fig. 6 presents regional mitigation and emission trade costs in 2020 and 2050. A numerical table with additional details is

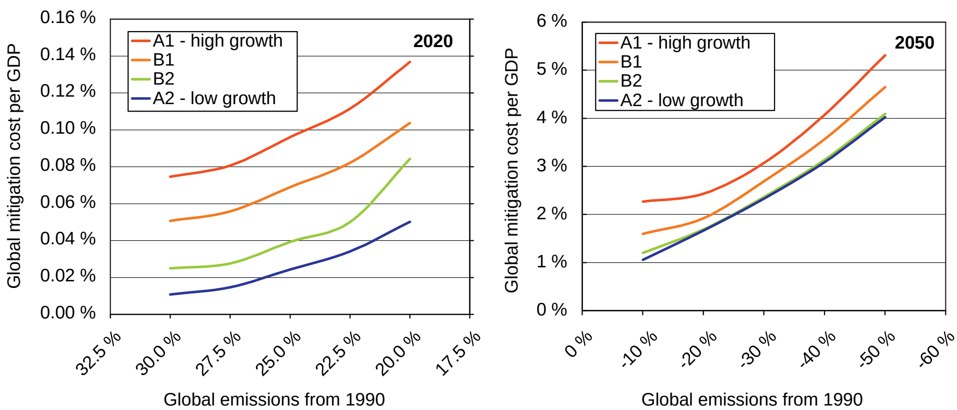


Fig. 4. Global mitigation costs relative to global GDP (*Y*-axis) in 2020 (left) and 2050 (right), with the different economic growth scenarios and emission reduction targets (relative to 1990 emissions, *X*-axis).

provided in Appendix A. Both effort sharing rules allocate costs for Annex I countries in 2020 (with the exclusion of Eastern Europe), costs around zero for more developed non-Annex I countries, and gains for least developed countries as a result of selling emission allowances. In 2050, Annex I countries, especially Australia and Russia (as a part of FSU) with the 485 ppm target, face relatively high costs. Also most non-Annex I countries face positive costs, and only India and Africa are able to gain financially from the effort sharing. The costs for Annex I regions are generally doubled with the 485 ppm target in 2050 compared to the 550 ppm target. A clear outlier from the overall pattern with all effort sharing rules is Middle East, the situation of which is analyzed briefly later.

For most regions the most important factor in the costs is allowance trade. Other factors include increased investment costs, reductions in fuel and operation costs and welfare losses as demand adjusts to higher energy prices. The volume of allowance trade can be substantial for some regions, especially in 2050 with the 485 ppm target when allowance prices are very high. The largest net seller in 2050 was India, which was able to sell allowances for from 1 Gt CO_2−eq (Triptych 485 ppm) to 4 Gt CO_2−eq (Multistage 550 ppm). Assuming a price of 500$/t, as an example, India would annually gain from 1% to over 10% of its baseline GDP from allowance sales in 2050, depending on the baseline. This would obviously have drastic impacts on the global economic system. For comparison, India's current account balance has been between -2.5% and 1.5% of GDP since 1980.

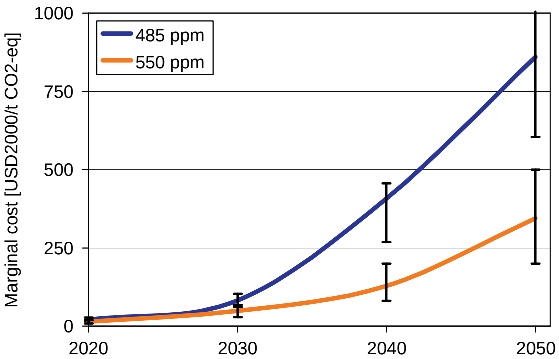


Fig. 5. Marginal costs of emission allowances ($\$_{2000}$/t CO_2-eq) in the scenarios. The error bars correspond to the range of values with four baseline scenarios.

As the Article 3.1 of the convention implies, the developed nations should take a lead in the mitigation effort. In order to assess the effort sharing in the light of the vertical equity principle, the regional mitigation costs were compared to the projected GDP per capita figures. Besides being equitable on a broad level, effort sharing should obviously be coherent by allocating similar costs for equally wealthy countries. An equitable and coherent effort sharing should then put the countries on an up-sloping line or a curve in the GDP per capita—mitigation cost plane. The slope of the curve should then be the subject of debate, that is, how much the more wealthy nations are seen to be responsible of taking on the costs.

In order to build more perspective, two very opposing effort sharing regimes are also portrayed in addition to Triptych and Multistage. An egalitarian approach, equal emissions per capita, has often been supported by developing countries. On the other hand, a grandfathering approach would be in line with the sovereign equity principle and favor the developed countries.

Fig. 7 portrays the regional mitigation costs against their GDP per capita projections, for 2020 and 2050 and both reduction targets. The figure includes also smoothed averages using Gaussian kernel smoothing to give better view on the overall equitability of each effort sharing regime.

Middle East, being an outlier from the overall pattern, was excluded from the kernel smoothing procedure. The mitigation costs in Middle East arise to a large extent from lower revenues from oil trade, resulting from a lower exports and oil price compared to the baseline scenarios, from 8% to 25% depending on the baseline and emission target, a phenomenon noted also by den Elzen et al. (2008b). Middle East is, however, a very heterogeneous group and the more wealthy oil-exporting countries, notably Saudi Arabia, Emirates, Kuwait and Qatar, constitute a relatively large share of both oil production and GDP in the region but only a small share of population, thus distorting the comparison between wealth and mitigation costs for Middle East.

As can be seen from Fig. 7, the differences between Triptych and Multistage in 2020 are relatively minor and fall between Per capita and Grandfathering approaches. The costs distribute equitably in the spirit of Article 3.1 with Triptych and Multistage approaches, with least developed regions having small negative costs, resulting from allowance sales, and developed regions having positive costs. While both approaches have a good coherence in costs vs. wealth, Triptych slightly outperforms Multistage in this sense. As was initially assumed, Per capita is very favorable to the least developed regions and Grandfathering for the developed.

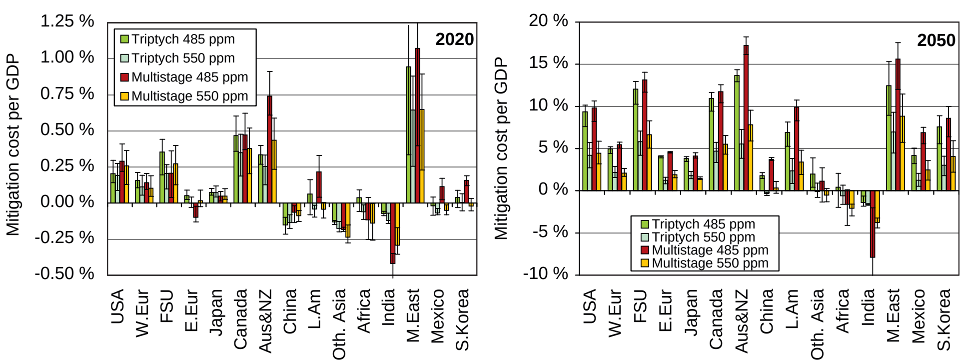


Fig. 6. Regional mitigation costs relative to their baseline GDP in 2020 (left) and 2050 (right). The error bars correspond to the range of values with four baseline scenarios.

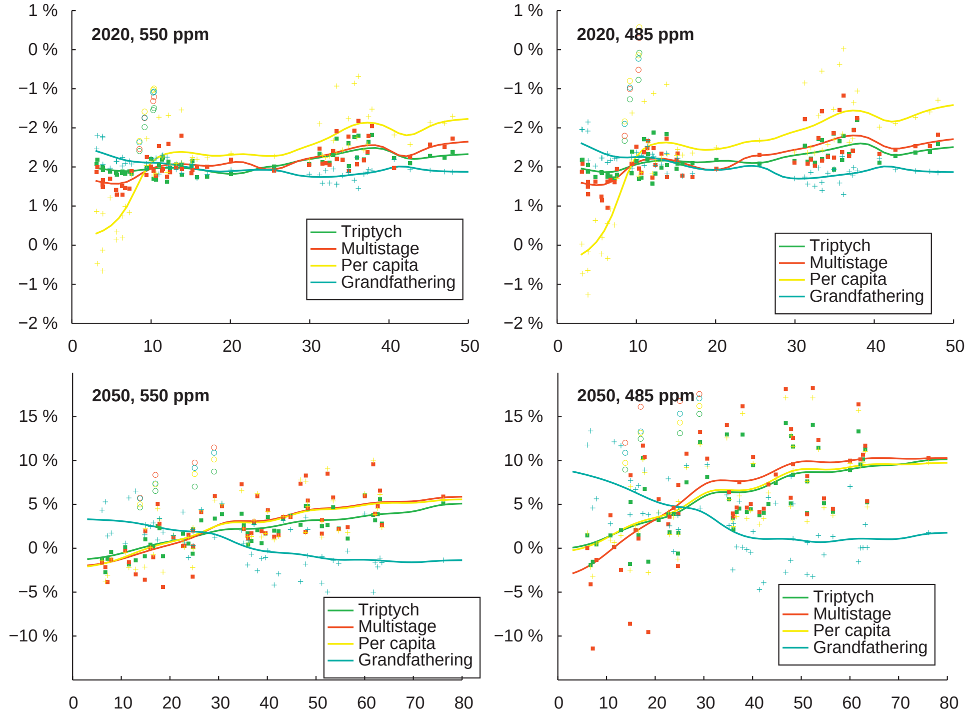


Fig. 7. Regional mitigation costs relative to GDP (*y*-axis) against regional GDP per capita (1000$$_{2000}$(PPP)/cap/a, *x*-axis), with four effort sharing rules and two mitigation targets in 2020 and 2050. Each dot or cross represents a single region with one of the four economic scenarios used. The solid lines are average values calculated with gaussian kernel smoothing. Middle East, marked with circles, has been excluded from the smoothed lines.

In 2050 the Triptych, Multistage and Per capita approaches produce very similar results on average, but Triptych exhibits some differences from the other two in the regional scale. As the emission converge to given emission per capita levels in the Multistage by 2050, the results between Multistage and Per capita approaches are very similar also in the regional level. However, Multistage is even more beneficial for least developed regions than the Per capita approach with the 485 ppm target, as some countries are still below the fourth stage threshold.

In the Triptych approach the sectoral emission converge to either "low" or "near-zero" levels (electricity, fossil fuel production and industry) or to given per capita levels (other sectors). Agriculture can also be included in the latter category, as the targets are defined as reductions from baseline emissions, which in turn are driven by population growth. This explains the similarity of Triptych and Per capita approaches, as a large share of the emissions allowances is allocated in per-capita term, especially with the 485 ppm target.

In terms of coherence Triptych again outperforms the other approaches clearly with the 550 ppm target in 2050, but not quite so with the 485 ppm target. This is again explained by the dominance of per-capita based sectoral targets, which is greater with the 485 ppm target. The coherence of Triptych is based on its ability to take into account the sectoral distribution of emissions in different countries, and thus also the countries' mitigation abilities. If the allowances are allocated mostly in per-capita terms, as with the 485 ppm target, coherence deteriorates.

3.4. Comparison to other studies

Comparison of the results to previous studies using different models reveals the importance of background assumptions used. Different studies can be distinguished with regard to the model used, baselines, available mitigation potentials, emission targets and effort sharing rules used.

Two different studies (van Vuuren et al., 2007; den Elzen et al., 2008b), using the IMAGE system in slightly different scenario settings, provide a good reference point. The emission levels, somewhat above 20 Gt CO_2−eq in 2050, fall between our 485 and 550 ppm targets. The marginal costs in den Elzen et al. (2008b) were between 125 and 270$/t$CO_2$−eq, which is generally lower than the range with our 550 ppm scenarios. The global costs were quite similar, around 1–2.5% of global GDP in 2050. The marginal and global costs in van Vuuren et al. (2007) with B2 fall into both of these ranges.

The differences in costs relative to the stringency of the emission target were attributed mostly to the assumptions on non-CO_2 mitigation and bioenergy supply potentials. The non-CO_2 potentials in the IMAGE model are based on an extension of the EMF-21 results (Lucas et al., 2007), and include rather optimistic estimates compared to those in the TIAM model. Also, bioenergy supply was limited to 500 EJ/a in den Elzen et al. (2008b), which is roughly four times larger than in our scenarios. Estimates both on bioenergy and non-CO_2 mitigation potentials are very uncertain, as was also acknowledged by den Elzen et al. (2008b). These assumptions have, however, a significant impact on the results, especially when deep emission reductions are assessed.

A comparison to Riahi et al. (2007), a mitigation scenario study using the MESSAGE model, is more difficult as the costs are reported only in terms of system costs and GDP losses, which is not directly translatable to the mitigation costs per GDP measure. However, an earlier study (Rao and Riahi, 2006) explores scenarios aiming at 4.5 and 3 W/m^2 radiative forcing targets by 2100 with multi-gas strategies. Although the radiative forcing targets equal those attained in the scenarios presented here, the emission profiles are very different with emissions exceeding 30 Gt CO_2-eq in 2050 in the MESSAGE scenarios and declining more later on. As a result, the marginal costs of emissions are also substantially lower in 2050, slightly above 100$/t CO_2-eq, but reach levels around 750$/t CO_2-eq by 2100.

The optimal profile of emission reductions is debatable, and cost-optimizing models such as TIMES and MESSAGE tend to postpone mitigation measures due to discounting if e.g. a radiative forcing or a temperature target is given instead of fixed annual caps. This can be also seen in a previous study with the TIAM modelling system (Syri et al., 2008), which investigated the optimal strategy for limiting global mean temperature increase below 2 °C by 2100. The optimization resulted with emissions around 30 Gt CO_2-eq in 2050, a level substantially higher than used in this study. However, as was also found by Syri et al. (2008), if stochastic optimization is used in the face of uncertainty in the climate sensitivity parameter, an optimal risk-hedging strategy would be to limit emissions to around 20 Gt CO_2-eq by 2050 in order to satisfy the 2 °C target. This result is therefore much in favor of targets lower than e.g. in Rao and Riahi (2006).

With regard to effort sharing, the results were compared to den Elzen et al. (2008b), which assessed Multistage and Contract and Converge effort sharing approaches. Even though having lower global mitigation costs, the patterns on how the cost is distributed is relatively similar to ours. Developed countries receive higher costs and least developed Sub-Saharan Africa and South Asia negative costs in 2050. Also, the countries under the former Soviet Union (FSU) region and Middle East fall outside the general pattern with higher costs, the former especially with Multistage effort sharing.

4. Challenges in effort sharing

Even if the effort can be shared in theory in a predetermined way, there are reasons why the economic burden might not distribute as planned. Perhaps the most evident is uncertainty in mitigation costs and the future price of allowances. In addition to this, the allowance market might not be perfect, which has been assumed in the analysis above, and this is analyzed in the case of transaction costs and imperfect participation to the market.

Also, the allocation of emission allowances is based on estimates on current emissions and sectoral mitigation potentials in the Triptych approach, but these parameters are not very well known. Although this uncertainty does not affect the analysis and methods used in this study as the allocation was taken as given, the allocation is obviously critical in defining the regional costs.

4.1. Imperfect markets

Two cases of market imperfections were considered to illustrate possible market-based hindrances for effort sharing. The first case introduces transaction costs in allowance trading, inhibiting the efficient functioning of markets. In the second case, a large net seller of allowances refuses to sell allowances to the market. Both cases were assessed in 2020 with the B2 growth scenario, 550 ppm mitigation target and Triptych effort sharing.

The introduction of transaction costs to the allowance market results with a situation where the sellers' and buyers' marginal abatement costs differ by the amount of the cost introduced. The cost might arise from numerous reasons, including imperfect information, market frictions or the faulting of the pricing mechanism, e.g. due to speculation. Some actors also might find it difficult or costly to trade in the market and monetary exchange rates might distort the efficiency of the market on a global scale. Also, volatile prices provide an incentive for risk averse hedging strategies that are somewhat costlier.

Due to the large number of potential sources, transaction costs are hard to quantify or forecast. To analyze its effect on the market, a quantification is, however, needed and as a rough guess a 10$/t CO_2-eq transaction cost was imposed to the markets. This can be seen as a moderate increase to the allowance price of 15$/t in 2020 in the setting without transaction costs. The cost reduced both the volume of emission trading by 20%, increased the costs of allowances by 23% (including the transaction cost), and doubled the global mitigation costs.

In the other case considered, a large net seller was assumed to refrain from trading its allowances. This can be conceptually contrasted from a scenario with limited participation in the overall mitigation effort, which has been analyzed previously e.g. by Edmonds et al. (2008). Even though all countries might comply with quantitative emission targets, there exists a risk that they will not participate in the allowance market in an efficient manner. China was chosen for this role for illustrative purposes, as it was the largest net seller of allowances in 2020 with Triptych effort sharing. It is also a large country holding slightly over 20% of all allowances with the Triptych allocation and might also hold relevant market power in practice.

In theory, a country cannot gain financially by restricting its allowance trading. Such action can be however easily justified. China faced some 40% increase in electricity prices and 90% increase in coal use costs when engaged with the global allowance markets in 2020. Coal and electricity make up over half of China's total final energy consumption in the baseline and over 80% in industry. Therefore, major political pressure might emerge against participating in the emissions trade if residents and companies were faced with steep increases in energy prices and were not compensated with the revenues from selling the allowances. Solutions to this dilemma might include using some of the emission trade revenues to subsidize clean energy production or consumption or a fragmented distribution of allowances to different actors in the allowance market.

On the global level, the setting resulted in one-third higher price for emission allowances compared to the basic setting, and a doubling of global mitigation costs. In contrast a surplus, though small, of allowances in China rendered their price to zero. In this scenario China loses its revenues from emissions trading but gains slightly on energy prices. Even though the total cost is slightly less than in the baseline, it is—as theory suggests—higher than in the case where China is selling its allowances.

4.2. Uncertainties

Uncertainties relevant for effort sharing arise from the baseline scenario, direct mitigation costs (technological and resource uncertainties) and allowance prices. Of these, the first was—to some extent—included in the analysis above with four baselines.

Technological and resource uncertainties affect in a simplified sense the marginal abatement curve (MAC) of a country. The effect on effort sharing is might be, however, small, as most technologies affect all countries. Then, a change in the costs or potential of a given technology affects effort sharing with countries that are more dependent on that technology than other countries. Such findings have been presented by den Elzen et al. (2005), where a second set of MAC's in the FAIR model raised uniformly the costs of all regions, although den Elzen et al. (2008b) noted that a specific technology's cost, CCS's in their case, might affect some countries more. The marginal mitigation cost is, however, also the basis for the price of allowances.

Allowance prices might also carry additional uncertainty due to market imperfections as was suggested in Section 4.1. The uncertainty in future allowance prices has important implications on the attainability of equitable effort sharing. The allowances have to be allocated to the countries in advance, and their value can be observed only later on.

As the price varies from 20% to 50% around the average between the scenarios with different baselines, and as the allowance trade might constitute a large share of region's mitigation costs, the price variability might affect the regional mitigation costs to a large extent. As the allowance trade costs are second order results from the model, they are more uncertain than most other results presented. However, with a given effort sharing regime, the amount of allowances a country buys or sells is relatively stable across the scenarios. In contrast, the price is very dependent on the background growth scenario. Uncertainties on marginal mitigation costs are in turn much larger for the more ambitious 485 ppm mitigation scenario, in which more unconventional measures have to be taken in order to reach the emission target.

4.3. Estimates of current emissions

Inventories or statistics on current emissions are far from perfect and subject to uncertainties, especially in the case of developing countries. Several organizations are providing emission estimates. Parties to the UN-FCCC are obliged to report their emission inventories, for Annex I parties annually and for the developing countries on a less frequent basis. The IEA publishes a global emission inventory from fuel combustion based on the energy statistics it gathers, supplemented with non-combustion emission estimates from the Emission Database for Global Atmospheric Research (EDGAR). Also, US-EPA has estimated global non-CO_2 emissions.

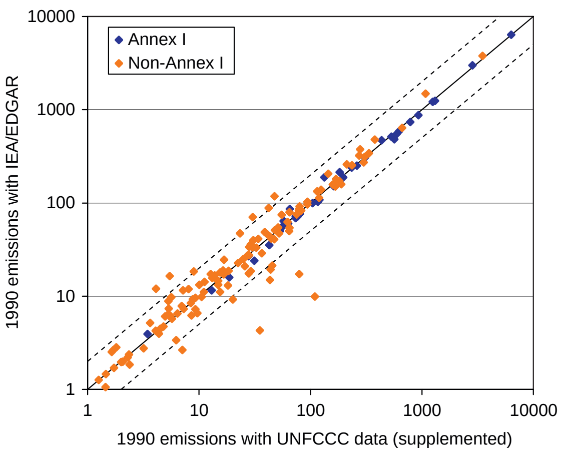


Fig. 8. Emission estimates for 1990 for different countries (Mt CO_2-eq, logarithmic scale) based on UN-FCCC (*X*-axis) or IEA/EDGAR (*Y*-axis). The dashed lines indicate points where one estimate is twice the magnitude of the other estimate.

The different datasets can exhibit considerable differences in their estimates. Fig. 8 presents emissions in the EVOC database with UN-FCCC and IEA/EDGAR-based data. Large deviations can be seen from the diagonal line, representing equal estimates between the sources, and for many individual countries the difference is over 100%, indicated by the dashed lines in the figure. As the effort sharing is based on these emission estimates, through sectoral projections in Triptych and emissions per capita in Multistage, the accuracy of emission estimates is material. Using different historical emission estimates might imply differences of several tens of percentage points on the allowances a country receives.

4.4. Assumptions behind the effort sharing

Obviously, effort sharing with the Triptych and Multistage approaches is dependent on the underlying assumption and parameter choices which define the allocation of emission allowances. Therefore a risk exists that if the parameters are inaccurate, the effort sharing can end up being erroneous.

This is especially problematic for the Triptych approach, as it is the more complicated one from the approaches assessed in this paper. The effort sharing with Triptych is based on assumptions on feasible mitigation potentials in each sector, which are in turn very uncertain in the very long term as noted in Section 3.1. Then, if the actual potentials in the future differ from those assumed, the emission allocation favors the countries, for which the mitigation potential has been underestimated.

During the study a notable difference in sectoral mitigation potential estimates—especially in agriculture—between EVOC and TIAM was noted, which prompted to a recalibration of EVOC to match the results from TIAM. Fig. 9 presents the results from EVOC for Triptych 550 ppm effort sharing in 2020 and 2050 before and after the recalibration. This recalibration had a large effect especially for certain countries. As an example, Australia received 66% more allowances in 2050 after the recalibration, reducing its economic burden substantially. A difference of this magnitude highlights clearly the importance of assumptions used in the effort sharing process.

5. Conclusions and discussion

This study has analyzed global effort sharing of climate change mitigation with Triptych and Multistage effort sharing rules and two mitigation scenarios aiming at -10% and -50% reductions from 1990 levels by 2050, leading to concentrations of 550 ppm CO_2-eq and 485 ppm CO_2-eq by 2100, respectively. Being simple and transparent, the EVOC tool of Ecofys GmbH was used for calculating Triptych and Multistage emission allocations, while and ETSAP-TIAM, a sophisticated but complex global energy system model of the TIMES family was used for creating the scenarios.

The available mitigation measures and their costs is crucial also for effort sharing, as the source distribution of emissions varies between countries and therefore regional mitigation potentials depend on the technological assumptions and resource estimates. Due to this, an explicit description of reduction measures undertaken in the scenarios was given. Most of the reductions were realized in electricity generation and industry. In other sectors numerous measures, however, mostly with limited potentials, were taken.

In the case of ambitious emission reductions, more unconventional measures have to be used. As many measures in transportation and agriculture were deemed to have limited mitigation potentials, reduced demand or substitution with lower

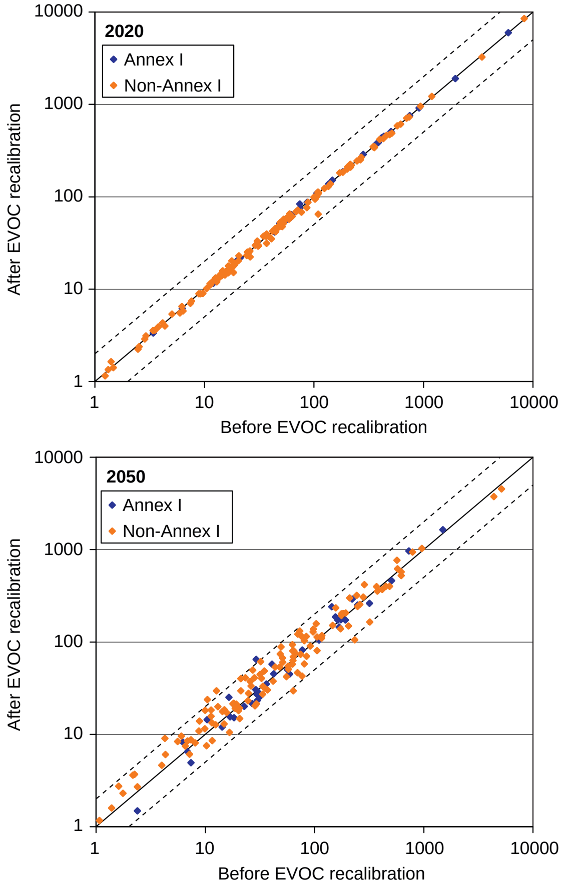


Fig. 9. Emission allocations (Mt CO_2-eq, logarithmic scale) with Triptych 550 ppm effort sharing before (*X*-axis) and after (*Y*-axis) EVOC recalibration in 2020 (top) and 2050 (bottom). The dashed lines indicate points where one estimate is twice the magnitude of the other estimate.

emitting alternatives was the only alternative for sufficient emission reductions. This was particularly the case with aviation, cattle and rice. The use of unconventional measures, however, also increases the uncertainty on mitigation costs and future allowance prices, rendering equitable effort sharing a challenging task.

The mitigation costs in the scenarios were relatively high compared to previous studies, reaching even 4–5% of global GDP with the 485 ppm target, by 2050. Also, the price of allowances was high, reaching even 1000$\$_{2000}$/t in 2050 with the 485 ppm target but being very dependent on the baseline scenario used. After accounting for differing reduction targets, the cost differences were identified to arise from less optimistic non-CO_2 mitigation potentials and the exclusion of afforestation options in this study. Although deforestation and afforestation are problematic for effort sharing due to the large uncertainties involved, they might be critical for reaching deep mitigation targets cost-efficiently.

Triptych and Multistage both allocate moderate reductions for Annex I and allow non-Annex I emissions to increase from 2000 levels by 2020. In 2050, Annex I faces very stringent targets around 80% from 2000 emissions, and only for the least developed non-Annex I regions the allowances exceeded their 2000 emissions. This is reflected also in mitigation costs with Annex I having positive costs and most or some non-Annex I regions having net gains due to allowance sales. Emission trading proved to be the most important single factor in the costs for most regions. The most extreme case was India in 2050, which was able to gain from 1% to over 10% of its baseline GDP from allowance sales.

A comparison between the economic burden the regions face and their abilities, by using GDP (PPP) per capita as a wealth measure, showed that both Triptych and Multistage produce equitable costs, although the balance of favoring the least developed and penalizing the most developed is obviously debatable. Overall, Triptych exhibited more moderate costs than Multistage for Annex I while still providing gains for non-Annex I, and might be thus be acceptable for both Annex I and non-Annex I. Triptych also exhibited higher coherence, i.e. the effort of individual regions varied less from the average. This highlights that an approach not taking into account the sectoral distribution of emissions and differing mitigation potentials can not adequately produce an equitable outcome. The coherence of Triptych did, however, degrade with the more stringent target, as the allocations are then mostly based on per-capita-based targets also with the Triptych approach.

Even if the effort can be shared equitably in theory, it might prove hard in practice. The future price of allowances varied considerably depending on the baseline, and studies with different models, and thus different assumptions, give even a wider range of possible price projections. A remark was also made on the data and assumptions behind effort sharing. Emissions estimates for especially non-Annex I are very uncertain, which makes effort sharing based on historical or projected emissions problematic. Also, if the effort sharing method specifies mitigation potentials in some form, as in the Triptych approach, these estimates have to be reliable, as was indicated by the Triptych recalibration experiment.

Given these uncertainties, fixing allowance allocations in the very long term might not be reasonable. As the mitigation costs cannot be accurately observed in reality, correcting distortions later on by reassessing the allocations would be challenging.

The analysis presented here has still some limitations. The partial equilibrium approach, while providing a detailed picture on the energy system, does not include any feedback effects from the rest of the economy. Effort sharing, especially in the extreme cases, might involve large wealth redistributions through allowance markets, affecting affluence levels and energy demand. Also, a high price of emissions is likely to induce structural change in the economy. Should the demand and production structures adjust to the cost of carbon, the mitigation costs then would be lower than reported here. With the TIAM model, the only possible adjustment is reduced demand, i.e. welfare loss, instead of e.g. demand substitution.

What was also not considered here, is the avoided damage costs from climate change through mitigation. Potential damage costs and adaptation capabilities vary largely between countries, and therefore should be also included in the analysis. This would, however, make the results unreliable due to the large uncertainties. Linking effort sharing to the funding of adaptation and technology transfer would still be reasonable, as all deal with transferring resources to the least developed and most vulnerable regions.

Last, the smooth operation of allowance markets and full participation of the parties is essential for cost-effectiveness.

Cases with transaction costs and limited participation both resulted with a doubling of global mitigation costs in 2020. Ensuring efficiency is, however, an issue of market design, but it might affect also effort sharing as the marginal costs are not necessarily equalized globally with inefficient markets.

Despite all these challenges, effort sharing is a necessity for the post-2012 climate policy. The negative costs for non-Annex I from the Triptych and Multistage, especially in the medium term, might provide a sufficient incentive for developing countries to accept binding targets. However, the gains are a result of wealth transfer from Annex I countries through allowance trading, the amount of which must be acceptable for Annex I countries. In this respect Triptych might provide a more balanced outcome of the two regimes assessed. It is yet good to bear in mind that the effort sharing will ultimately be a result of political negotiations. As said, there is no definitive answer to the equitable balance between costs and gains of different parties, but a quantified assessment of possible outcomes might aid the process considerably.

Acknowledgements

The funding from the Finnish Environmental Cluster Research Programme, the Ministry of Employment and the Economy, and VTT Technical Research Centre of Finland is acknowledged for the TIMES-Trip research project. Also, funding from Helsinki Institute of Science and Technology Studies (HIST) and VTT for writing the paper is appreciated.

Appendix A. Detailed results from effort sharing

Main quantitative results from effort sharing in the mitigation scenarios for each region is provided in Tables 2–5.

Appendix B. Supplementary data

Supplementary data associated with this article can be found in the online version at 10.1016/j.enpol.2009.11.055.

Table 2
Main outcomes of effort sharing with the 485 ppm-eq target in 2020—including GDP, baseline emissions, emissions after allowance trading, and allocations and mitigation costs with Triptych and Multistage effort sharing—with maximum and minimum values from the four baseline scenarios for each region.

	GDP (PPP) Bln. USD	Baseline emis. Gt CO_2−eq	Emissions Gt CO_2−eq	Triptych alloc. Gt CO_2−eq	Triptych cost Bln. USD	Multist. alloc. Gt CO_2−eq	Multist. cost Bln. USD
USA	15 000–16 000	7.9–8.4	6.6–7	5.1–5.7	21–48	4.4–4.9	32–66
W.Eur	14 000–15 000	4.7–5.1	3.7–4.2	2.9–3.2	14–33	3–3.3	13–32
FSU	3000–4100	3.4–3.9	2.8–3	2.3–2.5	5.8–17.4	2.6–2.8	1.2–15.0
E.Eur	1800–2600	0.88–0.99	0.72–0.79	0.73–0.78	0.4–2.3	0.89–0.95	−2.8–−0.6
Japan	4000–4500	1.3–1.4	1.1–1.1	0.86–0.94	2.4–4.5	0.89–1	0.7–3.6
Canada	1200–1400	0.75–0.79	0.59–0.64	0.37–0.41	4.4–8.2	0.36–0.4	4.5–8.5
Aus&NZ	820–920	0.74–0.74	0.57–0.62	0.52–0.56	2.2–3.7	0.34–0.38	5.0–8.4
China	12 000–17 000	6.6–7.3	5–5.4	6.8–7.7	−36–−11	6.2–7.2	−22–−2.7
L.Am	5300–6000	3–3.2	2.7–2.9	2.9–3.3	−4.4–9.7	2.3–2.8	2.2–19.8
Oth. Asia	5800–8100	3.2–3.9	2.8–3.3	3.4–3.7	−12.0–−7.4	3.6–3.9	−16–−11
Africa	4000–4800	3.2–3.4	2.8–2.9	2.9–3.2	−2.8–3.7	3–3.6	−12.0–1.6
India	5500–8500	2.9–3.6	2.2–2.5	2.7–3.1	−7.4–−3.1	3.8–4.5	−44–−19
M.East	3400–4000	2.8–3.2	2.5–2.8	2.3–2.5	12–58	2–2.3	14–65
Mexico	1700–1900	0.73–0.76	0.64–0.67	0.73–0.82	−1.4–0.8	0.57–0.7	0.4–3.3
S.Korea	1500–2100	0.74–1	0.59–0.75	0.56–0.62	−0.5–1.7	0.45–0.53	2.2–3.7

All values are on an annual basis, monetary values in USD2000.

Table 3
Main outcomes of effort sharing with the 550 ppm-eq target in 2020—including GDP, baseline emissions, emissions after allowance trading, and allocations and mitigation costs with Triptych and Multistage effort sharing—with maximum and minimum values from the four baseline scenarios for each region.

	GDP (PPP) Bln. USD	Baseline emis. Gt CO_2−eq	Emissions Gt CO_2−eq	Triptych alloc. Gt CO_2−eq	Triptych cost Bln. USD	Multist. alloc. Gt CO_2−eq	Multist. cost Bln. USD
USA	15 000–16 000	7.9–8.4	6.6–7	5.1–5.7	13–44	4.2–4.8	19–59
W.Eur	14 000–15 000	4.7–5.1	3.7–4.2	2.9–3.2	7–30	3–3.4	6–29
FSU	3000–4100	3.4–3.9	2.8–3	2.3–2.5	2.9–10.1	2.3–2.5	3.8–16.5
E.Eur	1800–2600	0.88–0.99	0.72–0.79	0.73–0.78	−0.5–1.1	0.78–0.83	−0.5–2.3
Japan	4000–4500	1.3–1.4	1.1–1.1	0.86–0.94	1.7–5.3	0.89–1	1.1–4.5
Canada	1200–1400	0.75–0.79	0.59–0.64	0.37–0.41	2.3–6.5	0.35–0.39	2.5–7.1
Aus&NZ	820–920	0.74–0.74	0.57–0.62	0.52–0.56	1.0–3.1	0.41–0.47	1.9–5.4
China	12 000–17 000	6.6–7.3	5–5.4	6.8–7.7	−29–−9	6.8–7.9	−21–−6.2
L.Am	5300–6000	3–3.2	2.7–2.9	2.9–3.3	−5.2–−0.9	3–3.7	−5.6–0.2
Oth. Asia	5800–8100	3.2–3.9	2.8–3.3	3.4–3.7	−15.6–−7.3	4.1–4.6	−22–−9
Africa	4000–4800	3.2–3.4	2.8–2.9	2.9–3.2	−5.4–−0.5	3.2–4	−12.3–−0.1
India	5500–8500	2.9–3.6	2.2–2.5	2.7–3.1	−11.7–−3.8	3.8–4.5	−30–−9
M.East	3400–4000	2.8–3.2	2.5–2.8	2.3–2.5	9–35	2.1–2.5	8–36
Mexico	1700–1900	0.73–0.76	0.64–0.67	0.73–0.82	−1.5–−0.7	0.72–0.88	−1.5–−0.2
S.Korea	1500–2100	0.74–1	0.59–0.75	0.56–0.62	−0.9–1.2	0.63–0.73	−1.0–0.6

All values are on an annual basis, monetary values in USD2000.

Table 4
Main outcomes of effort sharing with the 485 ppm-eq target in 2050—including GDP, baseline emissions, emissions after allowance trading, and allocations and mitigation costs with Triptych and Multistage effort sharing—with maximum and minimum values from the four baseline scenarios for each region.

	GDP (PPP) Bln. USD	Baseline emis. Gt CO_2-eq	Emissions Gt CO_2-eq	Triptych alloc. Gt CO_2-eq	Triptych cost Bln. USD	Multist. alloc. Gt CO_2-eq	Multist. cost Bln. USD
USA	22 000–30 000	8.8–11	2–2.4	0.56–0.63	1658–3052	0.39–0.56	1788–3094
W.Eur	18 000–27 000	5.2–6.4	1.1–1.4	0.58–0.66	787–1417	0.41–0.6	897–1450
FSU	6600–14 000	4.8–7.7	0.56–1.2	0.45–0.56	679–1819	0.34–0.44	774–1931
E.Eur	3100–6700	1.1–1.7	0.18–0.23	0.16–0.19	119–279	0.13–0.17	141–301
Japan	4900–6800	1.3–1.5	0.18–0.29	0.16–0.19	172–271	0.13–0.19	196–262
Canada	1800–2700	0.83–1.1	0.073–0.25	0.065–0.073	173–301	0.042–0.06	190–312
Aus&NZ	1100–1600	0.76–0.88	0.11–0.2	0.084–0.09	147–217	0.027–0.037	184–267
China	22 000–47 000	7.3–11	1.7–2.3	2.3–2.7	320–1005	1.6–1.9	822–1695
L.Am	12 000–18 000	4.8–6.5	1.3–1.6	1.3–1.6	672–1505	0.85–0.94	1057–1887
Oth. Asia	13 000–28 000	4.9–8.7	1.5–2.3	2.3–2.4	56–1107	2.4–2.7	−170–771
Africa	12 000–22 000	6.4–8.5	2–2.3	2.9–3.3	−260–453	3.2–4	−545–25
India	14 000–38 000	4.5–8.6	1.1–1.3	2.3–2.6	−455–−208	3.2–4.8	−2767–−778
M.East	8500–15 000	5.4–8.8	1.1–1.6	0.92–1.1	892–2349	0.54–0.76	1197–2696
Mexico	3900–6200	1.2–1.6	0.29–0.34	0.33–0.35	136–315	0.18–0.2	236–453
S.Korea	2800–5900	0.95–2.1	0.11–0.32	0.11–0.13	153–524	0.06–0.069	176–587

All values are on an annual basis, monetary values in USD2000.

Table 5
Main outcomes of effort sharing with the 550 ppm-eq target in 2050—including GDP, baseline emissions, emissions after allowance trading, and allocations and mitigation costs with Triptych and Multistage effort sharing—with maximum and minimum values from the four baseline scenarios for each region.

	GDP (PPP) Bln. USD	Baseline emis. Gt CO_2-eq	Emissions Gt CO_2-eq	Triptych alloc. Gt CO_2-eq	Triptych cost Bln. USD	Multist. alloc. Gt CO_2-eq	Multist. cost Bln. USD
USA	22 000–30 000	8.8–11	3.4–3.7	1.2–1.4	595–1721	0.9–1.1	699–1769
W.Eur	18 000–27 000	5.2–6.4	1.8–2	1.1–1.2	281–780	1.1–1.2	305–719
FSU	6600–14 000	4.8–7.7	1.8–2.1	0.87–1.1	278–1009	0.7–0.76	335–1179
E.Eur	3100–6700	1.1–1.7	0.35–0.38	0.37–0.4	26–107	0.26–0.3	47–161
Japan	4900–6800	1.3–1.5	0.37–0.48	0.29–0.31	70–154	0.32–0.37	68–113
Canada	1800–2700	0.83–1.1	0.36–0.39	0.15–0.19	61–153	0.1–0.11	79–175
Aus&NZ	1100–1600	0.76–0.88	0.25–0.38	0.17–0.21	44–118	0.091–0.1	67–156
China	22 000–47 000	7.3–11	3.1–3.7	4.5–5	−180–−44	3.7–4.8	−66–514
L.Am	12 000–18 000	4.8–6.5	2.5–3	2.3–3	110–708	2–2.3	245–887
Oth. Asia	13 000–28 000	4.9–8.7	2.6–3.4	3.7–4	−108–260	4–4.3	−212–73
Africa	12 000–22 000	6.4–8.5	3.4–3.5	4–4.8	−215–144	5–5.6	−655–−201
India	14 000–38 000	4.5–8.6	1.9–2.2	3.7–4.6	−649–−213	5.2–6.2	−1278–−524
M.East	8500–15 000	5.4–8.8	2.3–2.9	1.6–2.1	445–1426	1.1–1.4	572–1761
Mexico	3900–6200	1.2–1.6	0.49–0.63	0.65–0.71	22–129	0.47–0.55	54–223
S.Korea	2800–5900	0.95–2.1	0.35–0.54	0.26–0.4	49–241	0.17–0.2	60–348

All values are on an annual basis, monetary values in USD2000.

References

Böhringer, C., Welsch, H., 2004. Contraction and convergence of carbon emissions: an intertemporal multi-region cge analysis. Journal of Policy Modeling 26 (1), 21–39 doi:10.1016/j.polmod.2003.11.004.

DeAngelo, B.J., de la Chesnaye, F., Beach, R.H., Sommer, A., Murray, B.C., 2006. Methane and nitrous oxide mitigation in agriculture. The Energy Journal 27, 89–108.

den Elzen, M., Lucas, P., van Vuuren, D., 2005. Abatement costs of post-kyoto climate regimes. Energy Policy 33 (16), 2138–2151.

den Elzen, M.G., Berk, M., Lucas, P., Criqui, P., Kitous, A., 2006. Multi-stage: a rule-based evolution of future commitments under the climate change convention. International Environmental Agreements: Politics, Law and Economics 6 (1), 1–28.

den Elzen, M.G., Höhne, N., Brouns, B. Winkler, H., Ott, H.E., 2007. Differentiation of countries' future commitments in a post-2012 climate regime: an assessment of the south–north dialogue proposal. Environmental Science & Policy 10 (3), 185–203, doi:10.1016/j.envsci.2006.10.009.

den Elzen, M., Höhne, N., Moltmann, S., 2008a. The triptych approach revisited: a staged sectoral approach for climate mitigation. Energy Policy 36 (3), 1107–1124 doi:10.1016/j.enpol.2007.11.026.

den Elzen, M.G.J., Lucas, P.L., Van Vuuren, D.P., 2008b. Regional abatement action and costs under allocation schemes for emission allowances for achieving low CO_2-equivalent concentrations. Climatic Change 90 (3), 243–268.

Edmonds, J., Clarke, L., Lurz, J., Wise, M., 2008. Stabilizing CO_2 concentrations with incomplete international cooperation. Climate Policy 8 (4), 355–376 doi:10.3763/cpol.2007.0469.

FAO, 2009. FAOSTAT database. URL: faostat.fao.org.

Fisher, B., Nakicenovic, N., Alfsen, K., et al., 2007. Climate change 2007: mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Groenenberg, H., Phylipsen, D., Blok, K., 2001. Differentiating commitments world wide: global differentiation of GHG emissions reductions based on the triptych approach—a preliminary assessment. Energy Policy 29 (12), 1007–1030 doi:10.1016/S0301-4215(01)00027-1.

Höhne, N., Phylipsen, D., Moltmann, S., 2006. Factors underpinning future action. Technical Report, Ecofys GmbH.

IPCC, 2000. Special Report on Emissions Scenarios. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. In: Eggleston H.S., Buendia L., Miwa K., Ngara T., Tanabe K. (Eds). Prepared by the National Greenhouse Gas Inventories Programme. IGES, Japan.

Loulou, R., 2008. ETSAP-TIAM: the TIMES integrated assessment model part II: mathematical formulation. Computational Management Science 5 (1–2), 41–66.

Loulou, R., Labriet, M., 2008. ETSAP-TIAM: the TIMES integrated assessment model part I: model structure. Computational Management Science 5 (1–2), 7–40.

Loulou, R., Remme, U., Kanudia, A., Lehtilä, A., Goldstein, G., 2005a. Documentation for the times model. Technical Report DM 70046/ICC03080, IEA Energy Technology Systems Ananlysis Programme (ETSAP) 〈http://www.etsap.org/documentation.asp〉.

Loulou, R., Waaub, J.-P., Zaccour, G., 2005b. Energy and Environment. Springer, US (Chapter 11).

Lucas, P.L., van Vuuren, D.P., Olivier, J.G., den Elzen, M.G., 2007. Long-term reduction potential of non-CO_2 greenhouse gases. Environmental Science & Policy 10 (2), 85–103 doi:10.1016/j.envsci.2006.10.007.

Macintosh, A., Wallace, L., 2009. International aviation emissions to 2025: can emissions be stabilised without restricting demand? Energy Policy 37 (1), 264–273 doi:10.1016/j.enpol.2008.08.029.

Manne, A.S., Stephan, G., 2005. Global climate change and the equity-efficiency puzzle. Energy 30 (14), 2525–2536.

Miketa, A., Schrattenholzer, L., 2006. Equity implications of two burden-sharing rules for stabilizing greenhouse-gas concentrations. Energy Policy 34 (7), 877–891.

Persson, T.A., Azar, C., Lindgren, K., 2006. From production-based to consumption-based national emission inventories. Energy Policy 34 (14), 1889–1899.

Peterson, S., Klepper, G., 2007. Distribution matters—taxes vs. emissions trading in post kyoto climate regimes. Kiel Working Paper 1380, Kiel Institute for the World Economy.

Phylipsen, D., Höhne, N., Janzic, R., 2004. Implementing triptych 6.0—technical report. Technical Report.

Phylipsen, G.J.M., Bode, J.W., Blok, K., Merkus, H., Metz, B., 1998. A triptych sectoral approach to burden differentiation; ghg emissions in the European bubble. Energy Policy 26 (12), 929–943 doi:10.1016/S03014215(98)00036-6.

Rao, S., Riahi, K., 2006. The role of non-CO_2 greenhouse gases in climate change mitigation: long-term scenarios for the 21st century. The Energy Journal 27, 177–200.

Riahi, K., Grübler, A., Nakicenovic, N., 2007. Scenarios of long-term socio-economic and environmental development under climate stabilization. Technological Forecasting and Social Change 74 (7), 887–935.

Ringius, L., Torvanger, A., Holtsmark, B., 1998. Can multi-criteria rules fairly distribute climate burdens? OECD results from three burden sharing rules. Energy Policy 26 (10), 777–793.

Russ, P., Ciscar, J. C., Szabo, L., 2005. Analysis of post-2012 climate policy scenarios with limited participation. Technical Report.

Syri, S., Lehtila, A., Ekholm, T., Savolainen, I., Holttinen, H., Peltola, E., 2008. Global energy and emissions scenarios for effective climate change mitigation—deterministic and stochastic scenarios with the TIAM model. International Journal of Greenhouse Gas Control 2 (2), 274–285.

Vaillancourt, K., Waaub, J., 2004. Equity in international greenhouse gases abatement scenarios: a multicriteria approach. European Journal of Operational Research 153 (2), 489–505.

van Vuuren, D.P., den Elzen, M.G.J., Lucas, P.L., Eickhout, B., Strengers, B.J., van Ruijven, B., Wonink, S., van Houdt, R., 2007. Stabilizing greenhouse gas concentrations at low levels: an assessment of reduction strategies and costs. Climatic Change 81 (2), 119–159 doi:10.1007/s10584-006-9172-9.

	Series title and number **VTT Science 11**

Title	**Assessing the uncertainties of climate policies and mitigation measures** **Viewpoints on biofuel production, grid electricity consumption and differentiation of emission reduction commitments**
Author(s)	Sampo Soimakallio
Abstract	Ambitious climate change mitigation requires the implementation of effective and equitable climate policy and GHG emission reduction measures. The objective of this study was to explore the significance of the uncertainties related to GHG emission reduction measures and policies by providing viewpoints on biofuels production, grid electricity consumption and differentiation of emission reduction commitments between countries and country groups. Life cycle assessment (LCA) and macro-level scenario analysis through top-down and bottom-up modelling and cost-effectiveness analysis (CEA) were used as methods. The uncertainties were propagated in a statistical way through parameter variation, scenario analysis and stochastic modelling. This study showed that, in determining GHG emissions at product or process level, there are significant uncertainties due to parameters such as nitrous oxide emissions from soil, soil carbon changes and emissions from electricity production; and due to methodological choices related to the spatial and temporal system boundary setting and selection of allocation methods. Furthermore, the uncertainties due to modelling may be of central importance. For example, when accounting for biomass-based carbon emissions to and sequestration from the atmosphere, consideration of the temporal dimension is critical. The outcomes in differentiation of GHG emission reduction commitments between countries and country groups are critically influenced by the quality of data and criteria applied. In both LCA and effort sharing, the major issues are equitable attribution of emissions and emission allowances on the one hand and capturing consequences of measures and policies on the other. As LCA and system level top-down and bottom-up modelling results are increasingly used to justify various decisions by different stakeholders such as policy-makers and consumers, harmonization of practices, transparency and the handling of uncertainties related to methodological choices, parameters and modelling must be improved in order to avoid conscious misuse and unintentional misunderstanding.
ISBN, ISSN	ISBN 978-951-38-7469-8 (soft back ed.) ISSN 2242-119X (soft back ed.) ISBN 978-951-38-7470-4 (URL: http://www.vtt.fi/publications/index.jsp) ISSN 2242-1203 (URL: http://www.vtt.fi/publications/index.jsp)
Date	August 2012
Language	English, Finnish abstract
Pages	78 p. + app. 80 p.
Name of the project	
Commissioned by	
Keywords	Greenhouse gas emission, biofuel, electricity, effort sharing, uncertainty
Publisher	VTT Technical Research Centre of Finland P.O. Box 1000, FI-02044 VTT, Finland, Tel. 020 722 111

	Julkaisun sarja ja numero **VTT Science 11**

Nimeke	**Ilmastopolitiikkatoimien ja päästövähennysten epävarmuuksien arviointi** **Näkemyksiä biopolttoaineiden tuotannosta, verkkosähkön kulutuksesta ja päästövähennysvelvoitteiden taakanjaosta**
Tekijä(t)	Sampo Soimakallio
Tiivistelmä	Kunnianhimoiset tavoitteet ilmastonmuutoksen hillitsemiseksi edellyttävät tehokkaiden ja oikeudenmukaisten ilmastopolitiikka- ja päästövähennystoimenpiteiden toteuttamista. Tämän tutkimuksen tavoitteena oli analysoida kasvihuonekaasupäästöjen vähentämiseen liittyvien keinojen ja politiikkatoimenpiteiden epävarmuuksia tarkastelemalla biopolttoaineiden tuotantoa ja verkkosähkön kulutusta sekä päästövähennysvelvoitteiden taakanjakoa maiden ja maaryhmien välillä. Menetelminä käytettiin elinkaariarviointia, makrotaloustason skenaarioanalyysia ja kustannustehokkuusanalyysia. Epävarmuuksia tarkasteltiin tilastollisten menetelmien avulla mm. parametrien oletuksia vaihtelemalla, skenaarioanalyysilla ja stokastisella mallintamisella. Tulokset osoittavat, että tuote- tai prosessitasolla biopolttoaineiden tuotannon ja verkkosähkön kulutuksen kasvihuonekaasupäästöihin liittyy merkittäviä epävarmuuksia, joita aiheutuu arvioinnissa käytettävistä parametrioletuksista, esimerkiksi maaperän typpioksiduulipäästöille ja hiilivaraston muutoksille sekä sähköntuotannon päästöille. Epävarmuuksia aiheutuu myös tarkastelujen rajauksista ja allokointikäytännöistä sekä mallinnukseen liittyvistä tekijöistä, kuten biomassan hiilen vapautumisen ja sitoutumisen välisen ajallisen esiintymisen käsittelemisestä. Maa- tai maaryhmätasolla päästövähennysvelvoitteiden taakanjaossa sovellettavat kriteerit ja tietopohja ovat kriittisiä tulosten kannalta. Sekä elinkaariarvioinnissa että taakanjaossa päästöjen ja päästövähennysvelvoitteiden oikeudenmukainen kohdentaminen ja kerrannaisvaikutusten arvioiminen ovat keskeisiä tekijöitä ja voivat edellyttää useiden erilaisten menetelmien käyttämistä. Elinkaariarvioinnin ja järjestelmätason mallinnuksen tuloksia käytetään enenevässä määrin erilaisten päätösten perusteena. Tarkoitushakuisen väärinkäytön ja tarkoituksettomien väärinymmärrysten välttämiseksi on erittäin tärkeää, että elinkaariarviointiin ja järjestelmätason mallinnukseen liittyviä käytäntöjä yhtenäistetään, tulosten ja oletusten läpinäkyvyyttä lisätään ja menetelmiin, parametreihin ja mallinnukseen liittyvien epävarmuuksien käsittelyä parannetaan.
ISBN, ISSN	ISBN 978-951-38-7469-8 (nid.) ISSN 2242-119X (nid.) ISBN 978-951-38-7470-4 (URL: http://www.vtt.fi/publications/index.jsp) ISSN 2242-1203 (URL: http://www.vtt.fi/publications/index.jsp)
Julkaisuaika	Elokuu 2012
Kieli	Englanti, suomenkielinen tiivistelmä
Sivumäärä	78 s. + liitt. 80 s.
Projektin nimi	
Toimeksiantajat	
Avainsanat	Greenhouse gas emission, biofuel, electricity, effort sharing, uncertainty
Julkaisija	VTT PL 1000, 02044 VTT, Puh. 020 722 111

Assessing the uncertainties of climate policies and mitigation measures

Viewpoints on biofuel production, grid electricity consumption and differentiation of emission reduction commitments

Climate change is the major, primarily environmental issue of our time, and the single greatest challenge facing environmental regulators. Anthropogenic greenhouse gas emissions have increased significantly from the pre-industrial times. The consumption of primary energy has doubled since the early 1970s, and electricity consumption has increased almost fourfold. Ambitious climate change mitigation requires rapid and extensive measures, especially in energy production and consumption, enabling deep cuts in the GHG emissions within the upcoming centuries.

By the end of 2011, the viewpoints of the major emitters concerning binding GHG emission reduction targets and effort sharing between countries, have been too diverge for a breakthrough in international climate negotiations. However, various climate policies are implemented actively, in particular in the European Union. The use of renewable energy sources and transportation biofuels are promoted with mandatory commitments. At the same time, the environmental performance of product systems, over the life cycle from cradle to grave, is being increasingly assessed to justify various decisions.

Differentiation of emission reduction commitments between countries is a value-based issue. The implications of effort sharing may strongly depend on the criteria applied. When assessing GHG emission performance of product systems, a number of assumptions are required. This dissertation explores the significance of uncertainties related to GHG emission reduction policies and measures. Viewpoints on biofuel production, grid electricity consumption and differentiation of emission reduction commitments are provided.

ISBN 978-951-38-7469-8 (soft back ed.)
ISBN 978-951-38-7470-4 (URL: http://www.vtt.fi/publications/index.jsp)
ISSN 2242-119X (soft back ed.)
ISSN 2242-1203 (URL: http://www.vtt.fi/publications/index.jsp)