Integrated assessment of the introduction of biofuels in the Danish Transport sector

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Summary

A Multidisciplinary International Scientific Conference on 'Energy, Environment and Health – Optimisation of Future Energy Systems' was held over three days from May 31 to June 2, 2010 at Carlsberg Academy, Copenhagen, Denmark.

The organisers were the three research centres, CEEH (Centre for Energy Environment and Health, www.ceek.dk), REBECa (Renewable Energy in the transport sector - using Biofuels as Energy Carriers, biofuels.dmu.dk), and CEESA (Coherent Energy and Environmental System Analysis, www.ceesa.dk).

The conference brought together researchers from the scientific communities: Atmospheric physics and chemistry, air pollution modelling, environmental sciences, energy systems, human health and environmental economy. The aim of the conference was related to future energy scenarios and the consequences for health, environment, climate change and economy, with a focus on interdisciplinary support systems for assessment of future energy production and consumption, including direct and indirect costs.

The objective of the conference was to enhance the collaboration between scientists from different research fields with the mission to establish a common framework for interdisciplinary based systems to support optimal planning of future energy production and consumption where both direct and indirect costs related to human health, the natural environment and climate change are considered.

The conference contributions were held within the following sessions:

Session 1: Energy System Modelling

Modelling of energy systems, including conventional and renewable energy sources. Modelling of energy demand side, e.g. electricity, heating and transport. Integration of e.g. wind energy, biomass and hydrogen in energy systems. Integration of demand side in energy system models - saving, efficiency improvement and flexible demand. Energy system optimisation models.

Session 2: Environmental and Health Impacts

Describe the link between air pollution and environmental and health impacts. Quantification of the impacts from different pollutants on respiratory and cardiovascular diseases as well as on terrestrial and marine eco-systems. To investigate the effect of the chemical composition on toxicological impact and the link to epidemiology. Quantification of the health impacts of pollutants on a macro-scale level based on statistical methods. Dose-response functions as well as critical loads and levels.

Session 3: Economic valuation

Economic valuation of health and environmental externalities. Cost of impacts from air pollution on human health and the natural environment, including e.g. valuation of statistical lives (VSL) and value of life-years (VOLY), Unit values applied for assessment of the damage costs of air pollution, valuation of morbidity and mortality effects, cost of climate change (what is the cost of one kg CO_2).

Session 4: Integrated modelling and Optimization

Integrated modelling of energy systems, air pollution, environmental and health impacts, economic valuation and climate change. Description of - and typical results from - integrated modelling frameworks

that include one or more of the above mentioned components. E.g. integration of energy systems and air pollution modelling, integration of air pollution, health impacts and cost, or climate changes and energy optimisation modelling systems, etc.

Session 5: Future scenarios for energy production and consumption

Future scenarios for energy production and consumption with respect to costs related to the natural environment and human health. Optimisation of future energy systems including direct as well as indirect costs. Recommended scenarios as basis for optimal planning of future energy systems and solutions based on the interdisciplinary approach.

The conference took place at the Carlsberg Academy, which is the former residence of the Danish Nobel Prize winner Professor Niels Bohr.

Organising Committee: Lise Frohn[1], Allan Gross[1], Kenneth Karlsson[2], Jørgen Brandt[1], Eigil Kaas[3] .

1) National Environment Research Institute, Aarhus University 2) Risø National Laboratory for Sustainable Energy - Technical University of Denmark, 3) Niels Bohr Institute, University of Copenhagen

Session 1: Environmental and Health Impacts

Keynote speaker: Professor Steffen Loft, University of Copenhagen, Denmark

Chair: Jørgen Brandt

Health Impact Assessment in a Danish context

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Introduction

Health impact assessment is a combination of procedures, methods and tools by which a policy, program or project may be judged as to its potential effects on the health of a population, and the distribution of those effects within the population (The Gothenburg consensus paper 1999).

A quantitative Health Impact Assessment Model is a computer based tool to investigate and quantify both the health impacts and their possible economic consequences of an intervention with impact on public health. The aim is to measure these effects by comparing exposure to one or more risk factors in a reference scenario with the exposure in a planned intervention scenario. The impact on mortality can thus be measured as the difference in years in life expectancy between the two scenarios, when all other parameters are unchanged. In a further step differences in life expectancy can be valuated and thus quantified according to an economic measure.

Today there are various models to quantify health impacts, but these are primarily constructed according to cohort effects and can only account for heterogeneity by using indirect impact fraction techniques, which cannot directly answer questions about geographical variation in either risk factors of outcomes. The models are also deterministic and can only estimate uncertainties indirectly.

General Description

The Health Impact Assessment in CEEH (Centre for Energy, Environment and Health) will be carried out with a new Health Impact Assessment Model currently under development. The model will use a Multi State Markov framework, taking three different branches of public health related research areas into account: Demography, Epidemiology and Health Economics.

Changes in the gender and age composition of the population have to be taken into account when modelling the impact on health in the population of changes in the future exposure from air pollution. The demographic development in the coming years will change the composition of the Danish population with a growing proportion of elderly people. The Health Impact Assessment model presented will be able to take such changes into account.

With the expected demographic trends as a reference point, changes in exposure to different risk factors – according to air pollution scenarios in CEEH – changes in morbidity and mortality measured as years of life with illness or years of life lost due to too early death will be estimated at the gender and age specific level for each year modelled, and thus specify at which time point in the prediction these changes takes place.

In order to quantify the resource use and cost related to changes in mortality and morbidity on a scale comparable to other costs in the CEEH, disease specific costs associated with changes in mortality and morbidity will be applied and will be aggregated to net present values at various points in time.

The model will be presented using trends in cigarette smoking in Denmark from 1973 to 2006 as an example of a population exposed to a risk changing behaviour with impact on morbidity and mortality over a course of 34 years. The cigarette smoking scenario is chosen for two reasons. Firstly, because knowledge of the health risks of cigarette smoking is well established and can thus be used to validate different aspects of the model, and secondly, because the outcomes in terms of associated illnesses and causes of death caused by cigarette smoking are much alike those associated with air pollution – the topic of interest in CEEH.

The version of the model presented includes cigarette smoking as exposure and lung cancer morbidity and mortality as well as death from all other causes as outcomes. The model starts in 1973 and ends in 2006. Relative risk data is taken from Prescott *et al*. (1998).

Results

In figure 1 below the result of a model run is shown, with life years gained as outcome, with a fictions intervention where the smoking cessation probability is increased by 0.02 each year in all age groups above 29. As can be seen such a development in smoking cessation would have lead to a marked gain of life years throughout the period from 1973 to 2006 with app. 22,500 (21,083; 23,361) for men and 16,500 (15,577; 17,621) for women years gained in 2006. This is equal to an extra increase in life expectancy of 0.96 (0.75; 1.15) years and 0.69 (0.52; 0.90) years for men and women respectively.

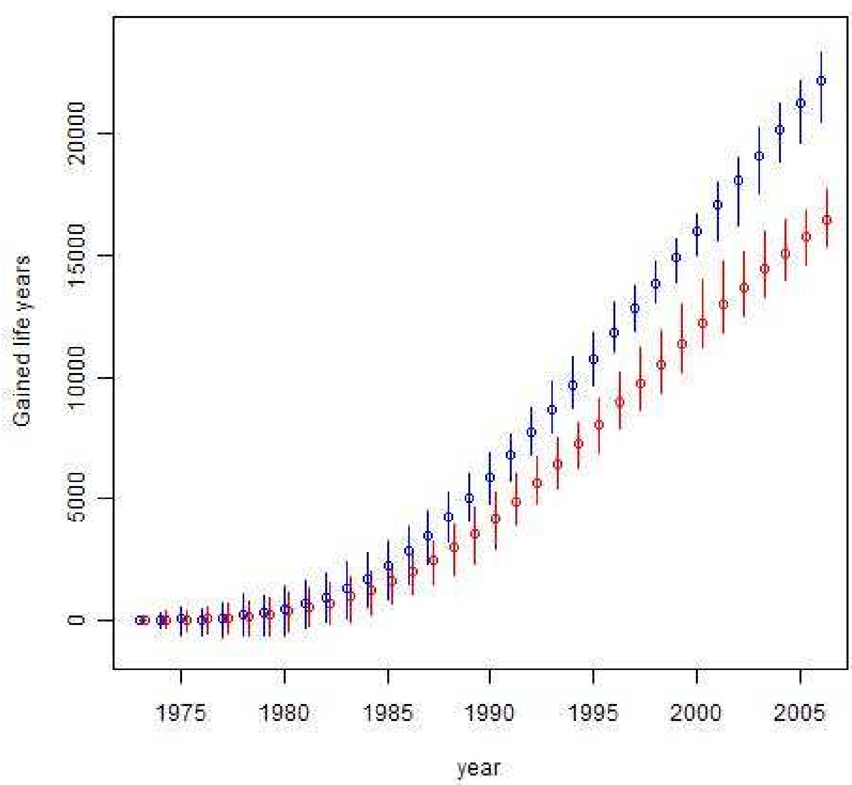

Figure 1: Number of gained life years for a fictions smoking cessation intervention, blue is men, red is women.

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Adverse effects of particles: Historical overview and future directions

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The first reports from of severe adverse health effects in large populations of combustion derived air pollution appeared last century with the 1930 Meuse Valley and the 1952 London fog episodes as the best known disasters.

Since then, population studies have evolved in size and methods and have gradually enhanced our knowledge of the adverse health effects caused by air pollution.

After the first smog episodes, it was commonly believed that extreme concentrations of smoke were necessary to cause an effect. Over the years, however, studies have demonstrated adverse effects at still lower concentrations of an increasing number of pollutants. Today, adverse effects including sudden death have been demonstrated even at everyday air pollution levels of metropolitan areas not generally considered very polluted. Accordingly recent studies have demonstrated positive effects of lowering air pollution even in little polluted areas. Continuous exposure for years to air pollution levels commonly encountered outside cities, major roads or industrial areas still causes shortening of human lives.

It has not been possible to establish a threshold below which no adverse effects can be observed. The relative risks associated with long-term exposure to air pollution have changed in the direction of stronger effects as new studies have appeared. An example is the 6% relative increase in all-cause mortality associated with a 10 µg/m3 increase in fine particulate matter in earlier studies (Pope *et al*., 2002) which in recent cohort studies have been proposed at 14% (Dockery *et al.*, 1993) and later 16% (Laden *et al.*, 2006).

The presentation will discuss examples of historical changes to concentrations of air pollutants causing adverse health effect. Major reasons why adverse effects apparently have been underestimated for years will be given. The most recent large studies in the field as well as studies underway in the near future will be presented.

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Worst Case study method to assess the environmental impact of amine emissions from a CO_2 capture plant

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Abstract

Use of amines is one of the leading technologies for post-combustion carbon capture from gas and coal-fired power plants. A CO_2 capture plant using amine technology will release amines (as gas and dissolved in droplets) to the air. These will undergo photo-oxidation and other reactions to form hundreds of different chemical compounds in the atmosphere after their release. The main aim of the current study is to estimate the potential adverse human health and environmental impacts given "worst case" assumptions on emission, dispersion and deposition of amines and their photo-oxidation products. We used a 40 x 40 km^2 study area in the vicinity of a planned 1 Mg/yr CO_2 capture facility at the gas-fired power plant at Mongstad, western Norway. We assumed that the plant would release 40 t/yr monoethanol amine (MEA) and 5 t/yr diethyl amine. With respect to inhalation exposure, the recommended risk threshold for N-nitrosodiethylamine (0.02 ng/m^3 in air; US EPA) was exceeded in the 40 x 40 km^2 study region. Drinking water standards for nitrosamines would be exceeded by about a factor of 3. MEA concentrations would exceed toxicity limits for aquatic organisms also by about a factor of 3. The "worst case" conditions may be different at other sites because the geographic location and the local meteorology have a large influence on both the atmospheric dispersion of pollutants and the local exposure of the population and the environment. Additional toxicity studies and field experiments are necessary to investigate biodegradation and retention of the compounds in soil and water before final conclusions can be drawn with respect to the maximum allowable emissions of amines and their oxidation products from CO_2 capture plants. The "worst case" approach can be applied to other emitted air pollutants.

1. Introduction

Post-combustion CO_2 capture has been proposed for two Norwegian gas-fired power plants (Kårstø, Mongstad) as a measure to reduce CO_2 emissions to the atmosphere. The most commonly used capture method is amine scrubbing. A CO_2 capture plant using amine technology will release amines (as gas and liquid) to air. In sunlight, amines undergo reactions with atmospheric oxidants involving oxidized nitrogen compounds (photo-oxidation) to form compounds such as nitrosamines, nitramines, and amides (Pitts et al., 1978). Nitrosamines are of particular concern, as they are toxic and carcinogenic to humans at extremely low levels (e.g. Reh et al., 2000). A recent screening project concluded that photo-oxidation of amines in the atmosphere produces compounds which based on reviewed toxicity data appear to be harmful to both humans and the local ecosystem (Knudsen et al., 2009).

Monoethanol amine (MEA, 2-aminoethanol) is the most widely studied solvent for the removal of CO_2 from flue gases. This study thus focuses on MEA, but in principle the methodology is applicable to other amines. A CO_2-capture plant that removes 1 Mt CO_2 per year from flue gas may emit 1-4 ppmv amines (NEV, 2006), which corresponds to the amount of 40-160 t/yr. In this study we assume that a blend of two amines, MEA and diethyl amine (DEYA), a secondary amine, is used for CO_2 capture and that maximum emission of 40 t/yr MEA and of 5 t/yr DEYA occurs due to volatilisation of the two amines from the scrubbing solvent.

Atmospheric dispersion modelling can be used to quantify the link between load (emission to air) and the resulting concentrations in air and flux in wet and dry deposition. The “worst case” approach here follows the precautionary principle and sets the most severe toxicological effect (lowest concentration at which an undesirable effect occurs) in relation to the expected maximum emission. By this approach it is possible to rank the hazard risk of the different chemical compounds and to prioritize the problematic compounds accordingly.

2. Methodology

The toxicology of generic amines and their possible photo-oxidation products was recently reviewed based on existing literature (Knudsen et al., 2008). Safety limits for the various compounds in air and in deposition were derived (Fig. 1, box 2) (Knudsen et al., 2008). Safety limits refer to the upper limit of the respective compounds in air and in deposition in order to avoid harmful effects to human health or to ecosystems. We used one year (2007) of synoptic meteorological data for Norway as input for the dispersion calculations with the dispersion model The Air Pollution Model (TAPM) developed by CSIRO, Australia (Hurley et al., 2005; TAPM 2009) (Fig. 1, box 3). Monthly average and 8-hourly maximum air concentration fields together with dry and wet deposition fields were obtained (Fig. 1, box 4). The maximum concentration and deposition flux in the study grid were compared to the pre-defined safety limits of the respective toxic compounds. Maximum tolerable emission rates were obtained by a back calculation procedure in which emissions from the plant were scaled until the safety limits in either air or deposition were reached (Fig. 1, box 5). An increase of the emission leading to concentrations and/or deposition fluxes beyond the critical level would then imply exceedence of the safety limit for a compound with negative impact on ecosystems and human health.

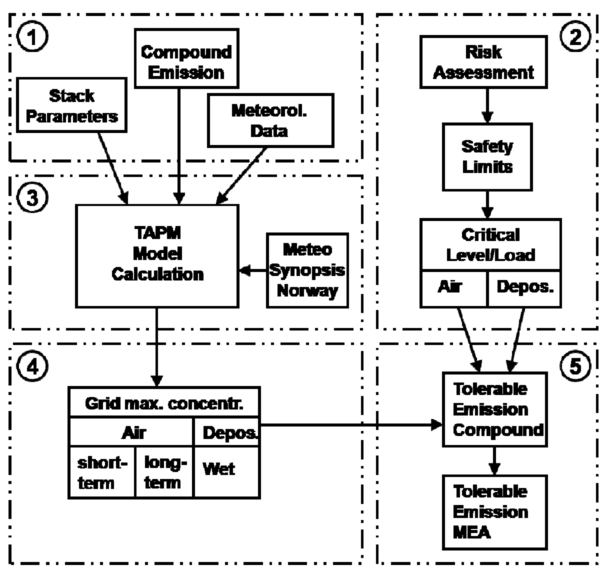

Figure 1: Flow scheme of the “worst case” method: 1) Emission input and local meteorological input; 2) determination of critical loads and levels for inhalation exposure, drinking water and aquatic environment; 3) dispersion calculations; 4) maximum air concentration and wet deposition fluxes in the grid obtained from the model run; 5) determination of tolerable emission amounts for the respective compounds and MEA.

For practical reasons, instantaneous conversion of the parent amines (MEA and DEYA) into photo-oxidation products was assumed to occur directly at the stack. Photo-oxidation products were grouped in chemical classes and the emissions were calculated as fraction of the parent amine emission using fixed chemistry formation yields. Atmospheric formation yields of MEA oxidation products were based on results from recent chamber experiments (Nielsen et al., 2010): 1% nitramines, 50% formamide, and 3% acetamide. Nitrosamines were not detected in the experiments. Formation yields of DEYA oxidation were adopted from an experimental study by Pitts et al. (1978), and are as follows: 3% nitrosamines, 32% nitramines, and 3% acetamide.

The dispersion model TAPM is an integrated model consisting of a prognostic meteorological module and a set of air quality modules (Hurley et al., 2005; TAPM 2009). In this application the meteorological module was nested three times, from an initial domain of 600 x 600 km (grid resolution of 15 km) down to a domain of 80 x 80 km (2 km resolution) centred on the Mongstad plant. Initial and boundary conditions for the outermost grid were taken from the six-hourly synoptic scale analyses derived by the LAPS or GASP models from the Australian Bureau of Meteorology. Surface boundary data, such as topography, land use and sea surface temperature were taken from the US Geological Survey, Earth Resources Observation Systems (EROS) Data Center Distributed Active Archive Center (EDC DAAC) and the US National Center for Atmospheric Research (NCAR). Emitted amines and their photo-oxidation products were assumed to be chemically inert in the atmosphere but undergo both wet and dry deposition processes. Deposition was treated in the same way as for sulphur dioxide.

3. Study Site

Mongstad (60°48’17’’ N, 5°01’50’’ E), Norway is located approximately 60 kilometres north of Bergen. Mongstad is situated at the coast and located only a few meters above sea level. The region is influenced by strong westerly winds from the Northern Atlantic for most of the year. To the east, the region is surrounded by a chain of hills and mountains up to 600m in elevation.

4. Results

Meteorological data of wind and precipitation for the year 2007 were retrieved for the few local meteorological stations within the study grid. Wind data were obtained with a six-hour resolution and rainfall data on a daily basis for the year 2007 from the Norwegian Meteorological Institute (met.no) (eKlima, http://sharki.oslo.dnmi.no). These data were used to evaluate prognostic wind and rain fields computed with TAPM. Monthly averaged wind speed as well as the seasonality of wind direction and speed at the met.no stations Takle, 30 kilometres northeast and Fedje, 18 kilometres were well reproduced by the dispersion model. Monthly averaged wind speed was underestimated by 10-50%. TAPM systematically overestimated the monthly rainfall amounts during the year 2007. The yearly rainfall pattern, however, was well captured by the model. The frequency of days with rain (rainfall amount >0.1 mm) in TAPM was about 20% higher than observed.

Based on TAPM calculations for the year 2007, yearly mean concentrations of MEA in air were below 0.05 $\mu g/m^3$ inside the 40×40 km^2 study domain (Figure 2). Monthly mean concentrations of MEA in air reached maximum values of 0.05-0.25 $\mu g/m^3$. Highest levels were 5-15 km to the north of Mongstad. In most months, the simulated plume also impacted the region south-east of Mongstad at a distance of 2-20 km, but monthly average concentrations were below 0.08 $\mu g/m^3$.

Inhalation Exposure. The inhalation exposure for MEA, nitrosamines (as group) and acetamide using the yearly average (MEA and acetamide) or 8-hourly average (nitrosamines) of air concentration assuming emission of 40 tonnes/yr MEA and 5 tonnes/yr DEYA was evaluated. The maximum yearly mean concentrations in air simulated in the study grid were 0.22 $\mu g/m^3$ MEA (safety limit: 10 $\mu g/m^3$) and 0.007 $\mu g/m^3$ acetamide (safety limit: 0.05 $\mu g/m^3$), respectively. The maximum 8-hourly average nitrosamine concentration was 0.011 $\mu g/m^3$ (safety limit: 1.0 $\mu g/m^3$). In the photo-oxidation of DEYA, N-nitrosodiethylamine (DEN) is formed. For DEN, the US Environmental Protection Agency (EPA) recommends a long-term risk threshold of 0.02 ng/m^3 in air, corresponding to a 10^{-6} lifetime cancer risk (US EPA, IRIS database: http://www.epa.gov/IRIS/subst/0042.htm). This criterion was exceeded by a factor of 32 in this study. These calculations assumed no chemical degradation of the compounds in air.

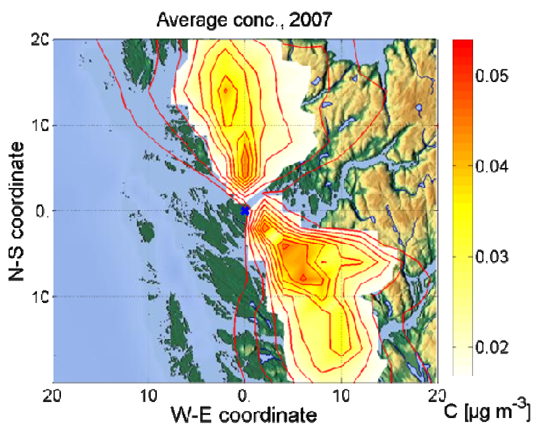

Figure 2: Simulated yearly mean MEA concentration in air ($\mu g/m^3$) given meteorological conditions of 2007 in the 40×40 km^2 domain around Mongstad (blue cross).

Drinking Water Quality. The critical nitrosamine load in the precipitation to Norwegian lakes with respect to drinking water quality is 7 ng/l. This corresponds to a critical annual deposition flux of 0.013 mg/m^2 under worst case conditions. This flux is reached with emissions of DEYA of 1.9 tonnes per year (Table 1), assuming a conversion of 3% of the emitted diethyl amine into nitrosamines. The assumed emissions of DEYA of 5 tonnes/yr exceed this value by about a factor of 3. The value was based on the predicted maximum annual wet deposition flux inside the study grid. The high wet deposition flux was only reached within a limited area of the study grid. With an instantaneous production yield of 1% nitramines from MEA and 32% from diethyl amine, maximum tolerable amine emissions of MEA and DEYA from the CO_2 capture facility are calculated to be 164 and 26 tonnes per year, respectively, to comply with the recommended drinking water threshold of 1 µg/l. Thus the drinking water criterion for nitramines was not exceeded (Table 1).

Compound	Safety Limit (ng/l)	Critical depostion flux (mg/m^2)	Deposition flux (mg/m^2)		Max. tolerable emission (t/yr)		Target
			Grid average	Grid max.	MEA	DEYA	
MEA	7,500	14.2	3.1	46	12		Aquatic algae/bacteria
Nitrosamines	7	0.013	0.01	0.03		1.9	Drinking water
	25	0.047	0.01	0.03		6.9	Aquatic algae/bacteria
Nitramine	1,000	1.9	0.22	0.83	164	26	Drinking water
	200	0.38	0.22	0.83	33	5.1	Aquatic Fish
Formamide	24,000	45.4	1.6	23	79		Aquatic Invertebrates

Table 1: Summary of maximum tolerable emission results for MEA and DEYA from the worst case studies. For the different compound classes, tolerable emissions of the parent amines MEA and DEYA were derived by converting the respective compound emission with the fractional yield.

Aquatic Environment. The safety limit of MEA with respect to aquatic organisms is 7.5 µg/l and corresponds to a critical annual deposition flux of 14.2 mg/m^2. Based on the maximum deposition flux found in the study grid, the maximum tolerable MEA emissions from a CO_2 capture plant would be 12 tonnes per year, about 3 times less than the assumed MEA emissions of 40 tonnes/yr (Table 1). Concentrations of formamide did not exceed the critical limit to aquatic organisms. Concentrations of nitramines were calculated to reach about 22% above the tolerable threshold for aquatic organisms and could cause chronic damage to fish.

5. Conclusions

The “worst case” approach evaluates the exposure to expected toxic compounds released from a CO_2 capture facility. Three exposure pathways were considered: 1) inhalation of air, 2) drinking water consumption, and 3) deposition to aquatic ecosystems. Among the expected substances, two compound groups are of particular concern due to their carcinogenic potential: nitrosamines and nitramines. Maximum wet deposition flux of nitrosamines (from diethyl amine) exceeded the safety limit for drinking water and maximum deposition flux of nitramines (from MEA) exceeded the safety limit for aquatic organisms. Toxicity to aquatic organisms is a major concern for the use of MEA since maximum tolerable MEA emissions are found to be only 12 t/yr. In this worst case study, we assumed that chemical compounds are stable in air, water and soil, with no degradation or loss during transport through each medium. Biodegradation of the compounds in soil and water is probably the highest uncertainty in the calculation of tolerable emissions. Though nitrosamines in water decay rapidly with light, they are more stable in drinking water systems and when mixed into deeper parts of lakes. Another uncertainty is the retention of amines in soils or sediments which might also reduce the concentrations in the lake, but was assumed to be negligible. Toxicity studies and field experiments are required before final conclusions can be drawn with respect to the maximum allowable emissions of amines and their oxidation products from CO_2 capture plants.

6. Acknowledgements

This work was supported in part by Statoil, Shell Technology Norway, Gassnova A/S, the Research Council of Norway (under the CLIMIT programme), the Norwegian Institute for Air Research NILU, and the Norwegian Institute for Water Research NIVA. We thank our colleagues at NILU, NIVA, the Chemistry Department of the University of Oslo, the Norwegian Institute for Public Health (FHI), and the Norwegian Institute for Nature Research (NINA) for fruitful collaboration in the research on the environmental effects of CO_2 capture by amines.

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Respiratory responses in atopic humans exposed to wood smoke

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Biomass combustion contributes widely to environmental air pollution. Experimental and epidemiological studies evaluating adverse health effects from ambient air pollution in relation to wood combustion indicate reliable relationship between particle exposure and increased incidences respiratory diseases.

The objective of this study was to examine whether short-term controlled exposure to wood smoke in atopic humans affects respiratory responses and markers of inflammation. An experimental set-up with a wood stove facility was used to generate wood smoke emissions inside a climate chamber. 20 nonsmoking atopic human participants with normal lung function and normal bronchial reactivity completed the study.

In this double-blinded study the participants were exposed for 3 h randomly to three different exposure levels at 2-week intervals. Exposures were clean filtered air (control exposure) and wood smoke with either a particulate matter (PM) concentration at 200 $\mu g/m^3$ or 400 $\mu g/m^3$. Health effects were evaluated at baseline and with follow-up measurement by changes in different lung function parameters and analysis of inflammatory cytokines in nasal lavages.

We found no significant effect of the exposure for any of the measured lung function parameters. The change in FEV_1/FVC between baseline and 30 min of exposure were Mean (C.I.) 0.8 (-0.5; 2.0) %, 0.4 (-0.8; 1.6) % and 2.1(-0.2; 4.4) % for clean filtered air, low and high wood-smoke particle level respectively. Neither for FEV_1, PEF nor for the other time points significant effects were found. However, significant differences between exposures in the levels of selected markers of inflammation (IL-8 and IL-12) were found.

This study indicates that the current experimentally generated wood smoke, dominated by PM, induces no acute decreased lung function in atopic humans, whereas inflammation level increased, indicating early systemic effects. Results of different lung function measurements and inflammatory markers will be presented and discussed at the conference.

Impacts of large-scale introduction of hydrogen in the road transport sector on urban air pollution and human exposure in Copenhagen

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1. Introduction

The vision of a renewable energy system with hydrogen as an important energy carrier requires a technological transformation of the present pre-dominantly fossil energy system. It involves the production of hydrogen with renewable energy sources, distribution of hydrogen in pipelines and e.g. new filling stations for cars, new technologies for use of hydrogen e.g. fuel cell cars etc. Such a hydrogen economy promises large environmental benefits for air pollution. Fuel cells combine oxygen from the air with hydrogen to produce electricity and only emit harmless water. When traditional combusting of fossil fuels is replaced the related emissions of pollutants are avoided. Reduced CO_2 emissions will reduce the green house effects. Furthermore, air quality problems in large cities and related public health problems will be diminished (Jacobsen et al., 2005). Ground level ozone will decrease and reduce related health problems and agricultural production losses. In addition, a reduction of environmental problems related to the deposition of air pollutants e.g. acidification and eutrofication will improve the natural environment for flora and fauna. However, there may also be drawbacks. The production, distribution and use of hydrogen will increase the leakage of hydrogen to the atmosphere. Recent research indicates that hydrogen may be involved in processes leading to increased depletion of the ozone layer and an increase in the green house effect (Tromp et al. 2003).

The aim of the project 'Environmental and Health Impact Assessment of Scenarios for Renewable Energy Systems with Hydrogen' (HYSCENE) is to improve modelling of the environmental impacts and related socio-cultural and welfare economic impacts of a proposed hydrogen/renewable energy system with focus on large-scale introduction of hydrogen as energy carrier in the road transport sector (http://hyscene.dmu.dk). This extended abstract will focus on the impacts on urban air pollution and human exposure.

2. Methodology

A baseline and a hydrogen scenario are defined for 2004, 2015, 2030 and 2050. Hydrogen is gradually introduced and in the hydrogen scenario it is envisaged that 75% of the total domestic transport energy demand in Denmark is covered by hydrogen in 2050.

Data for modelled fuel consumption in the baseline and H_2 scenario has been transformed into emissions. The calculation of emissions for the Danish road transport sector is based on a structure similar to the European COPERT emission model. For other mobile (national sea transport, domestic aviation, railways, military, non road working machinery, recreational craft) and stationary combustion sources, detailed activity based calculations are made following the European EMEP/CORINAIR guidelines for the historical situation. Appropriate forecast assumptions for fleet, activities and emission factors are used to predict the future emissions, also taking into account emission legislation already adopted. Emissions have been distributed geographically on a 1x1 km^2 grid for the road transport sector and on a 17x17 km^2 for other sources for air quality modelling (Winther et al. 2008).

The impacts on air quality on different geographic scales are assessed from regional to local level with the Greater Copenhagen Area as case study area. The Greater Copenhagen Area encompasses about 1.8 million people in a varied environment including the capital of Copenhagen, a number of middle-sized and small cities, and rural areas. On the regional scale, the hemispheric air chemistry transport model (Danish Eulerian Hemispheric Model - DEHM) is used to estimate air quality in Denmark (17x17 km^2) (Brandt et al. 2001; Frohn et al. 2002). Urban background concentrations are modelled on a detailed grid (1x1 km^2) with the Urban Background Model (UBM) (Berkowicz 2000b) and street concentrations are modelled with the Operational Street Pollution Model (OSPM)l (Berkowicz 2000a) for a large selection of streets in the case study area using the AirGIS system (Jensen et al. 2001;2009). Human exposure assessment is also carried out combining air pollution data with population data. Human exposure assessment is conducted on a 1x1 km^2 grid for the Greater Copenhagen Area. Very detailed population data (age, gender) is available based on the Central Person Registry (CPR) that includes population data for every single address in Denmark. Air quality assessment is also carried out for 138 busy streets in the capital of Copenhagen.

3 Baseline and H_2 Scenarios

The baseline and the H_2 scenarios include almost the same assumed total energy consumption in the Danish society over the years. The baseline scenario is based on the current official projections for the Danish energy sector. The H_2 scenario assumes a change into using H_2 as a primary energy carrier in the road transport sector.

The baseline scenario is extending from 2004 to 2050. This scenario assumes implementation of the current governmental energy plan (extending to the year 2030). For the years between 2030 and 2050, the development is established from linear extrapolation of the obtained trends between 2004 and 2030. The road transport accounts for 95% of the total domestic transport energy consumption in the baseline scenario.

The main objective of the H_2-scenario is to investigate the large-scale application of H_2 as an energy carrier in the Danish transport sector. In this scenario, it is assumed that the domestic road transport sector will obtain 75% coverage of H_2 as energy carrier by the year 2050. By 2050 the penetration of H_2 as energy carrier is assumed to be 100% in bus transport, 86% in passenger cars, and 65% in light and heavy duty vehicles. H_2 is projected to be introduced in the bus sector by 2008 and in the passenger cars as well as the light and heavy duty vehicles by 2015. The total H_2 share in the transport sector is assumed to 1% in 2015, increasing to 22% in 2030, and further to 75% by 2050. It is assumed that all H_2 driven vehicles will be based on fuel cell propulsion/electric engine systems, and not on internal combustion engines. The H_2 is assumed to be stored in compressed gas tanks and supplied in gaseous state. It is envisaged that refuelling facilities for vehicles and joint H_2 storages are established in appropriate accordance with the given role in overall power and heating system, and equivalent with the present refuelling infrastructure for conventional fuels.

Another assumption is that the transmission and distribution of H_2 will take place by means of the existing nationwide Danish natural gas pipe system. The loss of H_2 from fuel handling at filling stations, and leakage in transmission lines and central storage is quantified by application of leakage factors (0.05 in both cases, and the total H_2 mass is obtained from dividing with the lower heating value for H_2 (120 GJ/tons fuel). The total H_2 leakage is equivalent to about 10% of the consumption. The H_2 generation is established as part of the STREAM modelling of the electricity sector for the selected years. The STREAM model (Sustainable Technology Research and Energy Analysis Model) is a spreadsheet based energy system model covering the whole energy system. H_2 generation is assumed to take place by means of electrolysis based on electricity from the public grid in Denmark.

On the power supply side, the baseline scenario is more or less a continuation of the current power system in Denmark (the wind power share is increased from today's 20% to 30% in 2050). New more efficient power plants will replace old plants, but the combination of fuels used will still be the same. Wind power is increased during the scenario period and the future power production is mainly based on coal, gas and wind.

For the H_2-scenario, Denmark is assumed to be heading towards a fossil fuel free energy supply system. The driving factors for this development are here assumed to be high oil, gas and coal prices. For the H_2 scenario the Danish renewable resources useful for power production are mainly related to wind energy and biomass production. However, future energy resources may also include wave power and photovoltaic. The H_2 scenario stretches these resources to their limit and to keep up with the increasing power demand new coal fired power plants with carbon capture and sequestration (CCS) are build from 2015 as test plants and from 2030 as full scale and normal operating power plants.

4 Results and Discussion

4.1 Emissions and their geographic distribution

Total emissions from the baseline and the H_2 scenario are shown in Table 1 for CO_2-eq. (derived from CO_2, CH_4 and N_2O), NO_x, $PM_{2.5}$ and the hydrogen loss.

The largest emission reduction is seen for road transport due to the gradual shift to H_2. The non- exhaust $PM_{2.5}$ emissions arising from tyre, brake and road wear remain the same in the baseline and the H_2 scenario, and hence influences the total reduction in $PM_{2.5}$ emissions although $PM_{2.5}$ exhaust emissions are reduced drastically. For other sources than road transport, GHG emissions reductions are also expected, due to the assumed increase in the use of wind power and biomass in the H_2 scenario**.** However, the achieved emission reductions for other sources are more moderate than for road transport alone. The higher NO_x emission in the H_2 scenario compared to the baseline situation for other sources is due to an increased use of natural gas and biogas since they have very high NO_x emission factors due to combustion in gas engines. Further, the use of biomass does not bring down the NO_x emissions. The higher $PM_{2.5}$ emission for the H_2 scenario compared to the baseline is the large increase in biomass combustion especially in the residential sector associated with high emission factors. This illustrates the need for better emission control of these technologies if they are going to play important roles in future renewable energy scenarios. It may be considered an artefact that these emissions have not been lowered as emission control should be expected during the scenario period.

		CO_2-eq. (ktons)			NO_x (tons)			$PM_{2.5}$ (tons)				H_2(tons)
		Road	Other	Sum	Road	Other	Sum	Road exh.	Road non-exh.	Other	Sum	Sum
2004	Baseline	11953	53121	65074	66565	176401	242967	2876	842	25520	26363	0
2015	Baseline	12722	46816	59538	30842	117040	147882	1017	897	17770	18667	0
	H_2 scenario	12514	43364	55877	30144	120329	150473	1008	897	23382	24278	1420
	Reduction	209	3452	3661	697	-3289	-2591	9	0	-5611	-5611	-1420
	Red. (%)	2	7	6	2	-3	-2	1	0	-32	-30	
2030	Baseline	13910	45174	59084	14683	90273	104956	301	976	12070	13046	0
	H_2 scenario	10636	27653	38289	10250	92454	102704	231	976	16832	17808	17416
	Reduction	3274	17521	20795	4432	-2181	2251	70	0	-4762	-4762	-17416
	Red. (%)	24	39	35	30	-2	2	23	0	-39	-37	
2050	Baseline	15294	47358	62651	15239	93058	108297	301	1074	11121	12195	0
	H_2 scenario	3277	25186	28463	4242	95350	99593	103	1074	15753	16827	53824
	Reduction	12017	22172	34189	10997	-2293	8704	198	0	-4632	-4632	-53824
	Red. (%)	79	47	55	72	-2	8	66	0	-42	-38	

Table 1: CO_2-eq., NO_x, $PM_{2.5}$ and H_2 emission for baseline and H_2 scenario (Winther et al. 2008)

Figure 1 shows examples of geographic distribution of emissions for Denmark on a 17x17 km^2 grid and on a 1x1 km^2 grid for the road transport sector illustrated for the Greater Copenhagen Area. The increase in $PM_{2.5}$ emissions in the H_2 scenario is visible compared to the reference case mainly due to the increased use of biomass in the residential sector.

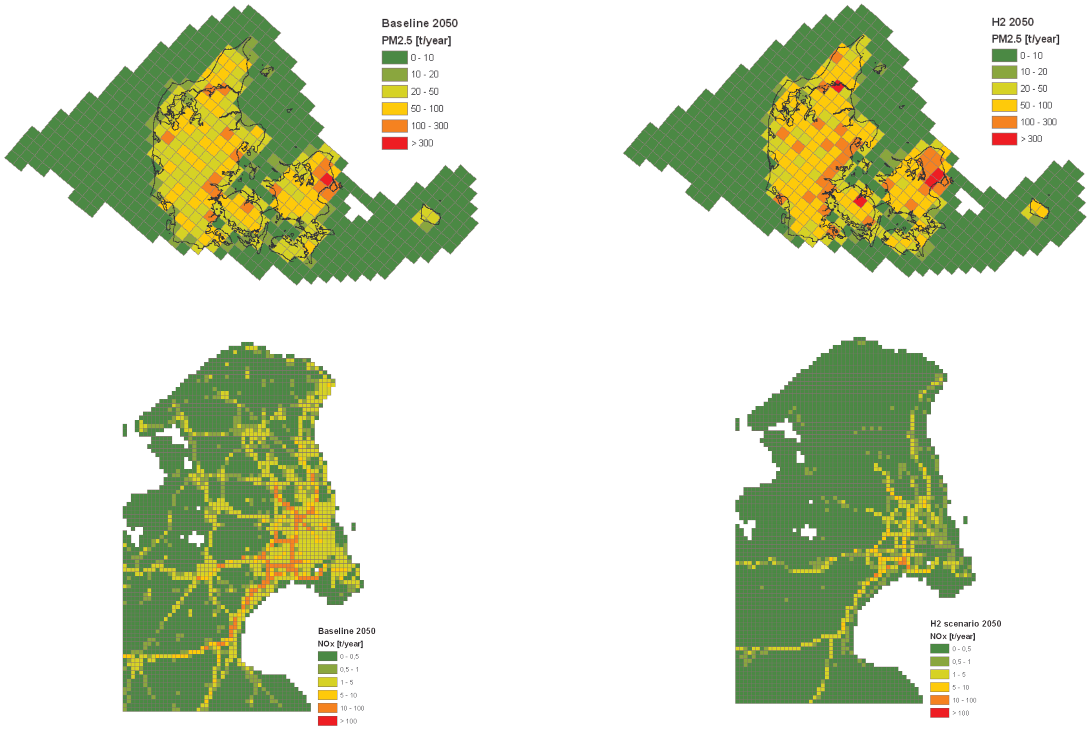

Figure 1: Upper: $PM_{2.5}$ emissions (17x17 km^2) for Denmark in 2050 (left: baseline and right: H_2 scenario). Lower: $PM_{2.5}$ emissions (1x1 km^2) for the Greater Copenhagen Area in 2050 (left: baseline and right: H_2 scenario) (Winther et al. 2008).

4.2 Air Quality Assessment

An example of primary $PM_{2.5}$ concentration difference between the baseline and the H_2 scenario at the regional level in Denmark based on the DEHM model is shown in Figure 2 (left). At the regional or background level it is seen that the concentration differences are very small between the baseline and the H_2 scenario as expected due to dilution and removal processes when pollutants are transported over long distances. Primary emitted $PM_{2.5}$ concentrations are slightly higher in the H_2 scenario due to higher $PM_{2.5}$ emissions due to increased use of biomass in the H_2 scenario where no mitigating emission control measures were assumed. An example of $PM_{2.5}$ concentration difference (only from primary emitted $PM_{2.5}$) between the baseline and the H_2 scenario at the urban background level in the Greater Copenhagen Area based on the UBM model is shown in Figure 2 (right). It is seen that the concentration differences are higher compared to the differences for the regional levels due to shorter dispersion distances. Primary $PM_{2.5}$ concentrations are slightly higher in the H_2 scenario due to the above reasons. To be able to compare with limit values for $PM_{2.5}$, the contribution from modelled secondary formed $PM_{2.5}$ concentrations (e.g. nitrates, sulphates) also has to be taken into account.

Although the H_2 scenario has slightly higher $PM_{2.5}$ concentrations in the regional and urban background, street concentrations of $PM_{2.5}$ are expected to be lower due to reductions in road emissions that contribute significantly to street concentrations (not shown).

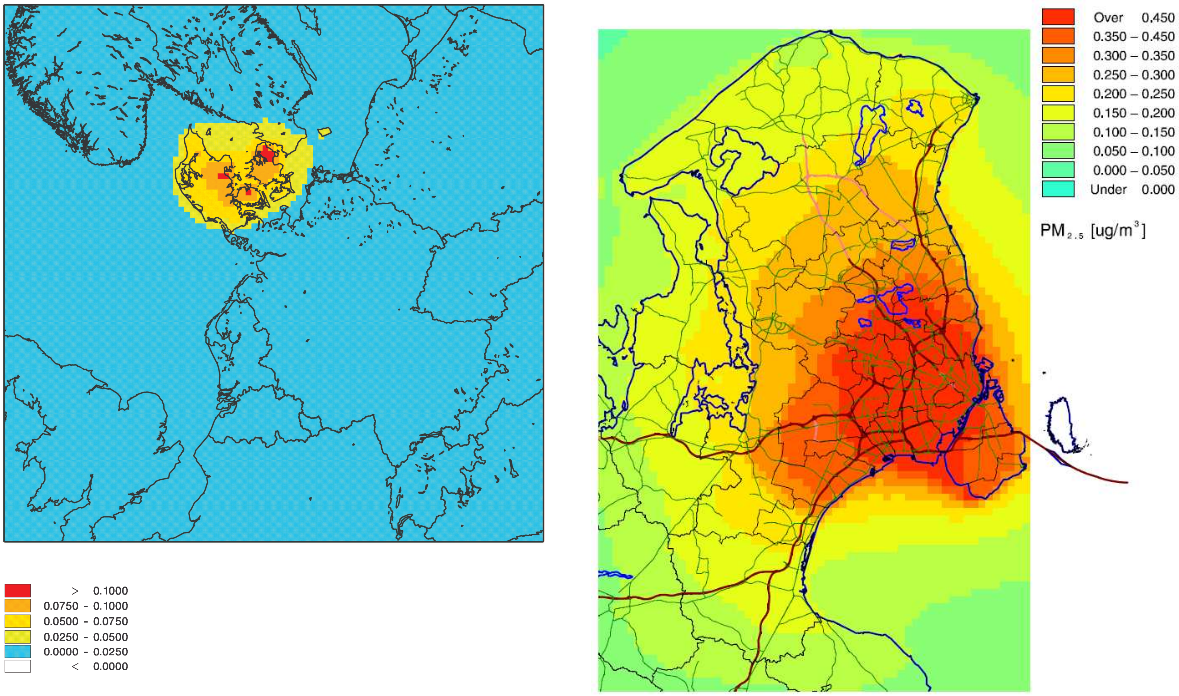

Figure 2 Left: Example of concentration difference between the baseline scenario and the H_2 scenario in µg/m^3. In this case in 2050 for primary emitted $PM_{2.5}$ modelled with the DEHM model on a 17x17 km^2 grid.

Right: Primary $PM_{2.5}$ concentration difference between the baseline and the H_2 scenario at the urban background level in the Greater Copenhagen Area based on the UBM model on a 1x1 km^2 grid.

Acknowledgement

The HYSCENE project is funded by the Programme Commission on Energy and Environment under the Danish Strategic Research Council (http://fist.dk/site/english). The project period was 2006-2008.

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The effect of wood smoke on vascular function and oxidative stress. An exposure study among 20 healthy subjects.

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Exposure to particulate matter is associated with the risk of cardiovascular events, possibly through endothelial dysfunction mediated by oxidative stress. However, the cardiovascular events caused by wood smoke particles have not yet been thoroughly investigated. This study evaluated the vascular effects in atopics exposed to wood smoke generated by wood-burning stoves in a full-scale chamber under controlled conditions.

Twenty-four non-smoking atopic subjects with normal lung function and normal bronchial reactivity were recruited and 20 completed the whole program. The subjects were exposed at rest in a randomized, double blinded, cross-over study for three episodes of 3½ hour exposures to 0, 200, or 400 mg/cm^3 wood smoke in a climate controlled chamber, at 2 week intervals.

Microvascular function (MVF) was assessed non-invasively by measuring digital peripheral artery tone following arm ischemia 5 hours after exposure. Before and at 0, 4.5, and 21 hr after exposure, blood samples were drawn and peripheral blood mononuclear cells (PBMC's) were isolated. The level of DNA damage in term of strand breaks and oxidized purines and pyrimidines were measured by the comet assay.

Results from an earlier study carried out by this group (Bräuner, 2008) show that decreasing indoor particle concentration was associated with improved MVF. Preliminary results from this study indicate that there is no effect on the MVF by exposure to wood smoke, but atopics had lower MVF compared to normal healthy subjects.

The level of strand breaks in PBMC's was elevated when the subjects had been exposed to higher levels of wood smoke particles.

In this study exposure to wood smoke is not associated with microvascular dysfunction in atopics, whereas DNA strand breaks increased in PMBC, indicating systemic oxidation effects. Gene and surface expression on PBMC is currently being analysed.

Reference:

Bräuner EV, Forchhammer L, Møller P, Barregard L, Gunnarsen L, Afshari A, Wåhlin P, Glasius G, Dragsted L, Basu S, Raaschou-Nielsen O, and Loft S. Indoor Particles Affect Endothelial Function in the Elderly: An Air Filtration-based Intervention Study. *American Journal of Respiratory Critical Care Medicine* ***177 (4): 419-25, 2008***

Emission consequences of introducing biofuels in the Danish road transport sector

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Introduction

Funded by the Programme Commission on Energy and Environment under the Danish Strategic Research Council, the multi-disciplinary integrated impact assessment project 'Renewable Energy in the transport sector using Biofuels as an Energy Carrier' (REBECa) is currently under implementation in Denmark. The aim of REBECa is to assess the impact on emissions, air quality and human health as well as resource and land-use change, and to consider economic and sociological aspects of the future use of biodiesel and bioethanol in Danish road transport. The project period is 2007–2010.

An important task of work package II (emission inventories) in REBECa is to estimate the fuel consumption and emissions for two fossil fuel based baseline scenarios for Danish road transport from 2004-2030, characterised by different traffic growth rates. For each of the baseline scenarios, two biofuel scenarios are considered with different penetration rates of biodiesel and bioethanol. Biofuel scenario 1 assumes an energy share of biodiesel and bio ethanol of 5.75 % in 2010, followed by a linear growth to 10 % in 2020, and with constant levels in the following years. In biofuel scenario 2 the biofuel share is also 5.75 % in 2010 and subsequently the biofuel share grows linearly to 25 % in 2030. For biodiesel full miscibility is assumed, whereas for bioethanol the definition is to add 5 % v/v mix of bioethanol in the standard gasoline fuel (E5), and let the surplus of ethanol available be used by FFV's (Flexible Fuel Vehicles) running on E85.

In the project, specific fuel consumption and emission calculations of CO_2, SO_2, NO_x, TSP, CO and VOC are made for the baseline year 2004, and the scenario years 2010, 2015, 2020, 2025 and 2030.

The purpose of the present paper is to describe the emission inventory and the calculated results. A short methodology description will be given in terms of fleet specific mileage data, baseline emission factors, biofuel emission difference functions and calculation method. In the results part, baseline emission results will be given in time-series. Further, comparisons will be made for the baseline and biofuel scenarios in the discrete scenario years in order to assess the emission impact of biofuel usage. Further, selected emission results are also displayed on GIS maps for Denmark.

Method

The mileage forecast used in the REBECa project is prepared by DTU Transport in Denmark. The mileage forecast which is based on an oil price of $65 pr barrel of oil (Fosgerau et al., 2007), is also used as an input to the Danish Infrastructure Commission (2008). Due to the very high oil prices in 2008 and the latest estimate of $100-120 pr barrel for the future oil price from IEA, an alternative mileage scenario for the REBECa project is also calculated by DTU Transport, based on an oil price of 100$ pr barrel. A documentation of the mileage forecast is given by Jensen and Winther (2009).

In order to make sufficiently detailed fuel consumption and emission estimates in REBECa, the DTU mileage figures must be grouped into vehicles with the same average fuel consumption and emission behaviour; the so-called layers. An internal model developed by NERI (Winther, 2008; Nielsen et al., 2009) uses a layer structure and calculation methodology similar to the model structure of the European emission calculation model COPERT. The layer splits are made according to fuel type, engine size/weight class and EU emission legislation levels. Figure 1 shows the layer split of DTU mileage forecast, aggregated from engine size (cars) and weight class (trucks) though.

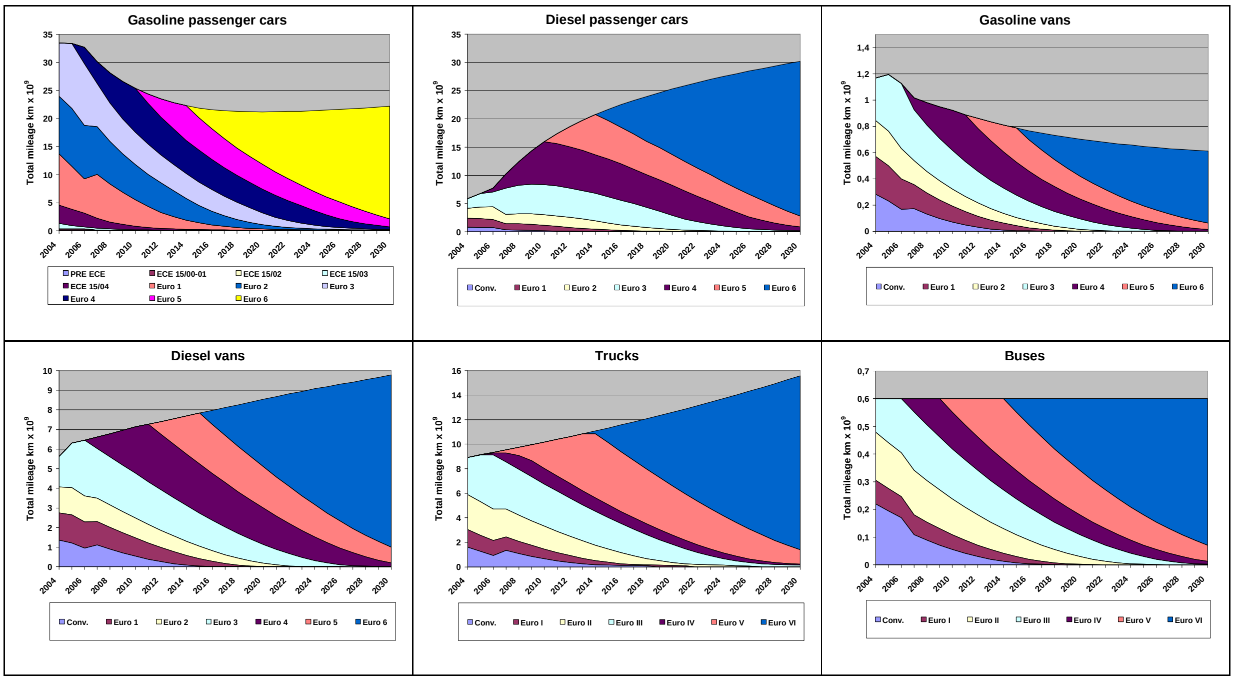

Figure 1: Layer distribution of total mileage pr vehicle type in 2004-2030.

Figure 2 presents the NO_x emission factors as an example for gasoline cars (year 2015, including cold start and catalyst wear) and diesel trucks, also weighted according to mileage pr road type.

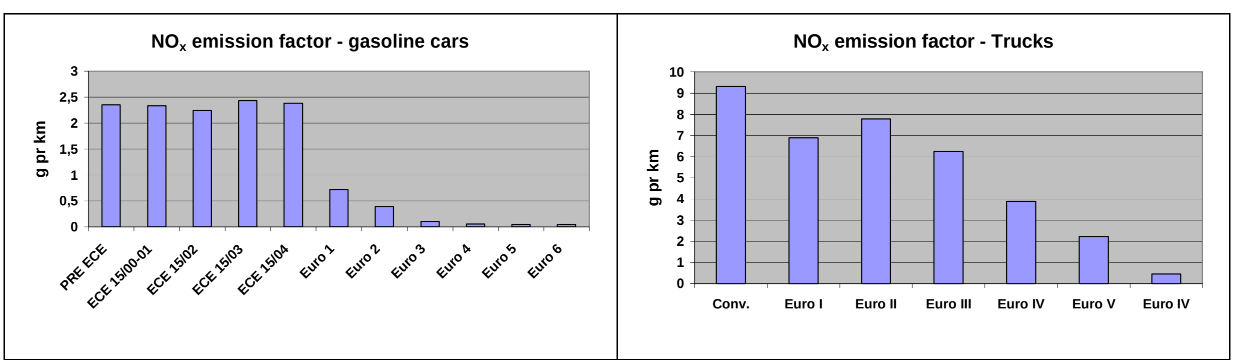

Figure 2: Layer specific NO_x emission factors for gasoline cars and diesel trucks.

For Euro 0-3 heavy-duty engines the changes in fuel consumption and NO_x, PM, CO and VOC emissions as a function of B%V, is based on the findings from EPA (2002). The data from the latter source is also used for the future Euro 6 engine technology, as assumed by Winther (2009). For Euro 4 and 5 engines, the experimental basis behind the curves is measurement results from McCormick et al. (2005). The fuel consumption and the Euro 0-3/Euro 4-5 emission curves for NO_x and PM are shown in Figure 3. For neat biodiesel, the CO*[VOC]* % emission changes are -48*[-67]* and -40*[-25]*, for Euro 0-3 and Euro 4-5, respectively.

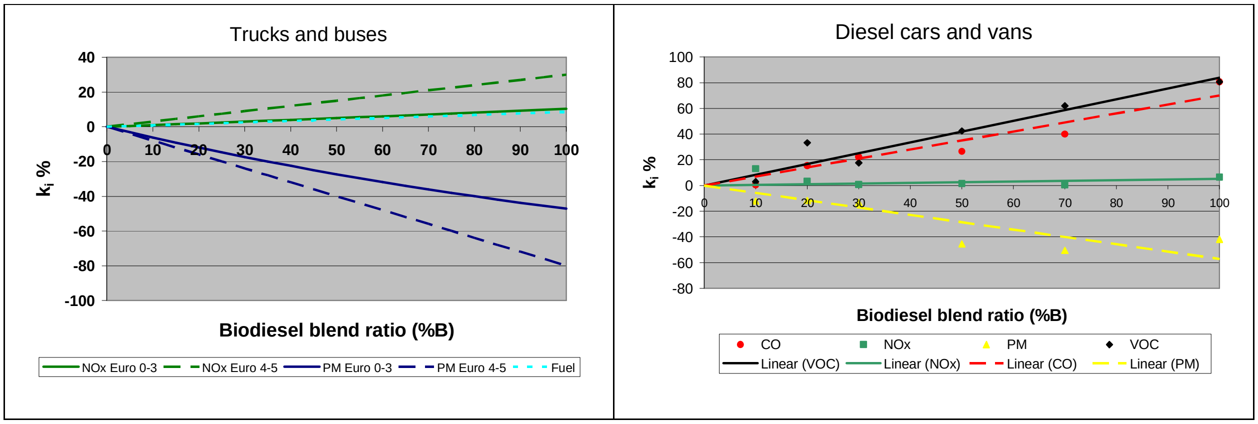

Figure 3: Fuel consumption/emission changes (function of B%) for heavy-duty engines and diesel cars/vans.

In the case of passenger cars and vans, average emission differences for B10, B20, B30, B50, B70 and B100 are calculated based on the results from four experimental studies, see Winther (2009). The emission differences expressed as linear functions are shown in Figure 8 for NO_x, CO, VOC and PM. For fuel consumption the relative changes were not derived explicitly for passenger cars and vans, due to lack of data. For these vehicle types, instead the general relations for heavy-duty vehicles are used. This decision is discussed in Winther (2009).

To characterise the energy consumption and emission factor differences between neat gasoline and E5 and E85, respectively, average differences are calculated from five European studies (three for E5, two for E85), see Winther (2010). In the experiments using E85 fuels, the base fuel was E5 since in Sweden the baseline fuel quality for petrol is predominantly E5. However, noting the small average differences between neat gasoline and E5 - and due to lack of experimental data for modern European cars using neat gasoline and E85 - the E5 vs. E85 differences are used in REBECa for the neat gasoline vs. E85 case as well. This decision is discussed in more details by Winther (2010).

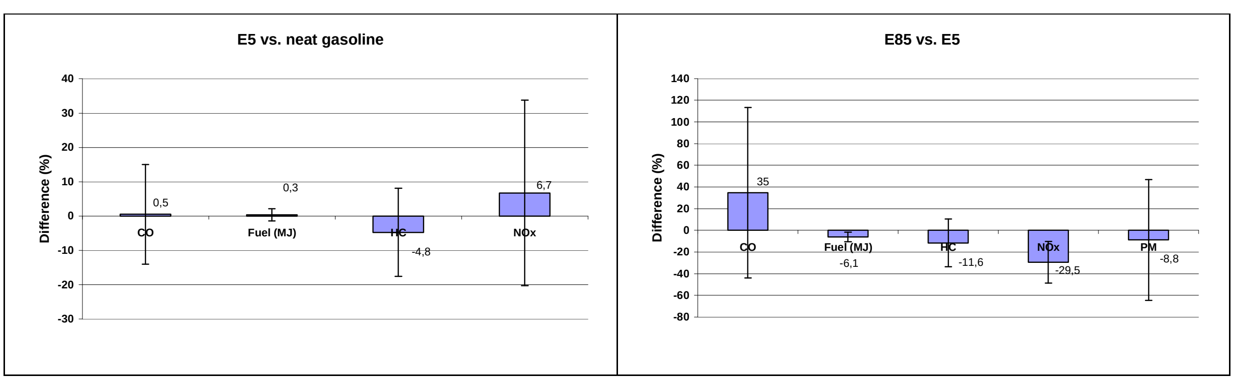

Figure 4: Fuel consumption and emission changes for neat gasoline and E5/E85 for gasoline cars and vans.

For each inventory year emission (and fuel consumption) results are calculated pr layer and road type. The procedure is to combine emission factors, emission change functions, number of vehicles, annual mileage levels and the relevant road-type shares:

$$E_{i,j,k,y} = emf_{i,j,k,y} \cdot (100 + k_i(B\%_V))/100 \cdot S_k \cdot N_{j,y} \cdot M_{j,y} \quad (1)$$

E = emission, emf = emission factor, ki = emission change function, i = emission component, y = inventory year, j = layer, S = road type share, k = road type.

For bioethanol it is assumed that in 2010 FFV's that belong to the most modern Euro layer for gasoline cars (Euro 4) uses the amount of ethanol not being used as E5 blends by gasoline vehicles as such. In 2015 the share of Euro 4 vehicles being FFV's is maintained, hence assuming approximately the same rate of scrapping of vehicles irrespective of technology. Further, the remaining ethanol surplus is assumed to be used by the most modern Euro classes in 2015 (Euro 5 and 6). This step wise ethanol allocation principle is used for the years 2020, 2025 and 2030 also.

Results

For the 65$ mileage forecast the calculated results are shown per vehicle category in Figure 10. The fuel consumption and CO_2 emissions increase by 43 % from 2004 to 2030. The emission increase is highest for heavy duty vehicles (trucks and buses) and vans, 51 % and 48 %, respectively, due to a larger traffic growth. For NO_x and PM, the emissions decrease by 81 % and 89 %, respectively. The NO_x and PM emissions decrease of 72 % and 83 %, respectively, for cars, are smaller than the total emission decreases, due to a gradually larger share of diesel cars expected in the future vehicle fleet. From 2004 to 2030 the CO and VOC emissions decrease by 82 and 78%, respectively (not shown).

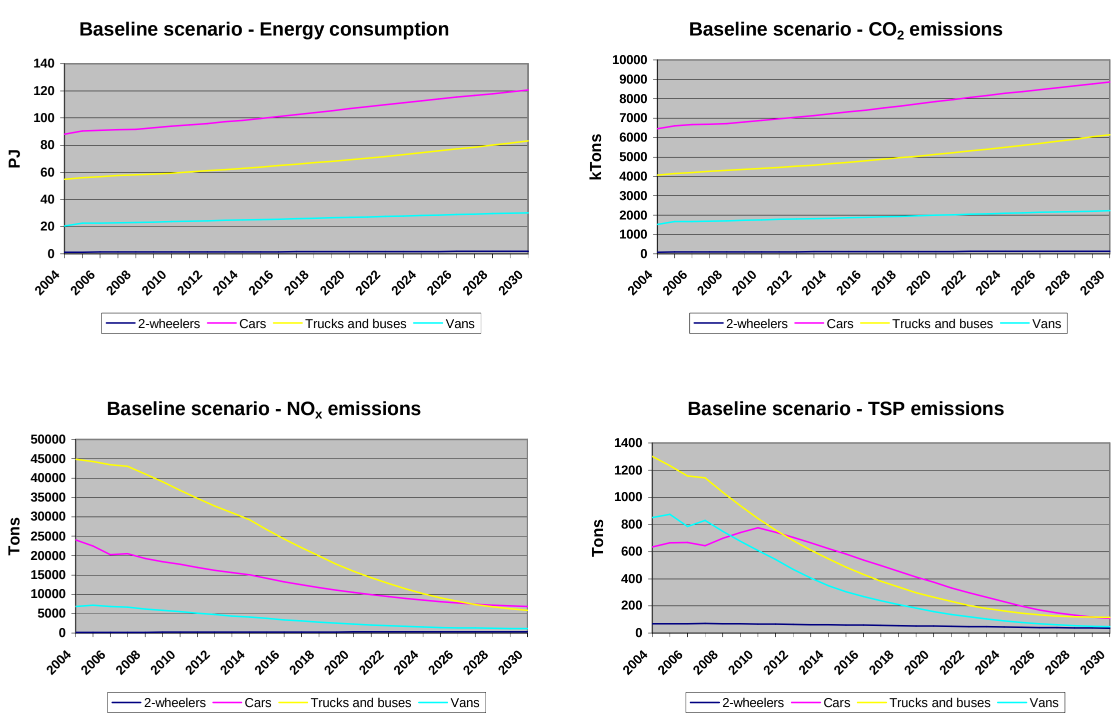

Figure 5: Total energy consumption and CO_2, NO_x and TSP emission baseline results pr vehicle type.

The emission consequences of using biofuel in road transport - even at blend ratios up to 25 % - are small. For NO_x and VOC the absolute differences between the baseline and biofuel scenarios are less than 3 %. For CO and exhaust PM the largest emission differences, 8 % and -13 %, respectively, occur between the baseline and biofuel scenario 2 in 2030, related to a biofuel share of 25 %. However, CO is of less environmental concern and if for PM the emission contribution coming from non exhaust is included in a total PM assessment, the emission differences between baseline and biofuel scenarios become considerably smaller.

		Mileage forecast: 65 $									Mileage forecast: 100 $								
Scen.	Year	En	NO_x	VOC	CO	CO_2	PM	TSP	PM_{10}	$PM_{2.5}$	En	NO_x	VOC	CO	CO_2	PM	TSP	PM_{10}	$PM_{2.5}$
							Exh.	Exh. + Non exh.								Exh.	Exh. + Non exh.		
S. 1	2010	-0,3	1,5	-2,5	0,6	-6,0	-3,6	-1,6	-2,0	-2,5	-0,3	1,4	-2,4	0,5	-6,0	-3,6	-1,6	-2,0	-2,5
	2015	-0,4	1,5	-1,7	1,0	-8,3	-4,9	-1,5	-2,0	-2,8	-0,4	1,5	-1,6	0,9	-8,3	-4,8	-1,5	-2,0	-2,8
	2020	-0,6	1,6	-0,7	2,0	-10,5	-6,0	-1,2	-1,7	-2,5	-0,6	1,6	-0,7	1,9	-10,5	-6,0	-1,2	-1,7	-2,5
	2025	-0,6	1,5	-0,1	2,6	-10,5	-5,6	-0,7	-1,0	-1,5	-0,6	1,5	-0,1	2,5	-10,5	-5,6	-0,7	-0,9	-1,5
	2030	-0,6	1,2	0,1	3,0	-10,5	-5,2	-0,4	-0,6	-1,0	-0,6	1,2	0,1	2,9	-10,5	-5,1	-0,4	-0,6	-1,0
S. 2	2010	-0,3	1,5	-2,5	0,6	-6,0	-3,6	-1,6	-2,0	-2,5	-0,3	1,4	-2,4	0,5	-6,0	-3,6	-1,6	-2,0	-2,5
	2015	-0,6	1,8	-1,7	1,4	-11,1	-6,5	-2,1	-2,7	-3,7	-0,6	1,8	-1,6	1,3	-11,1	-6,5	-2,1	-2,7	-3,7
	2020	-0,9	2,2	-0,4	3,2	-16,2	-9,1	-1,9	-2,6	-3,9	-0,9	2,2	-0,4	3,0	-16,2	-9,1	-1,9	-2,6	-3,8
	2025	-1,3	2,5	1,1	5,8	-21,2	-11,2	-1,3	-1,9	-3,0	-1,3	2,5	1,0	5,5	-21,2	-11,1	-1,3	-1,9	-3,0
	2030	-1,6	2,0	2,3	8,3	-26,2	-12,7	-0,9	-1,4	-2,3	-1,6	2,0	2,2	8,0	-26,1	-12,4	-0,9	-1,4	-2,3

Table 1: Fuel consumption and emission differences (%) between baseline and biofuel scenario 1 and 2.

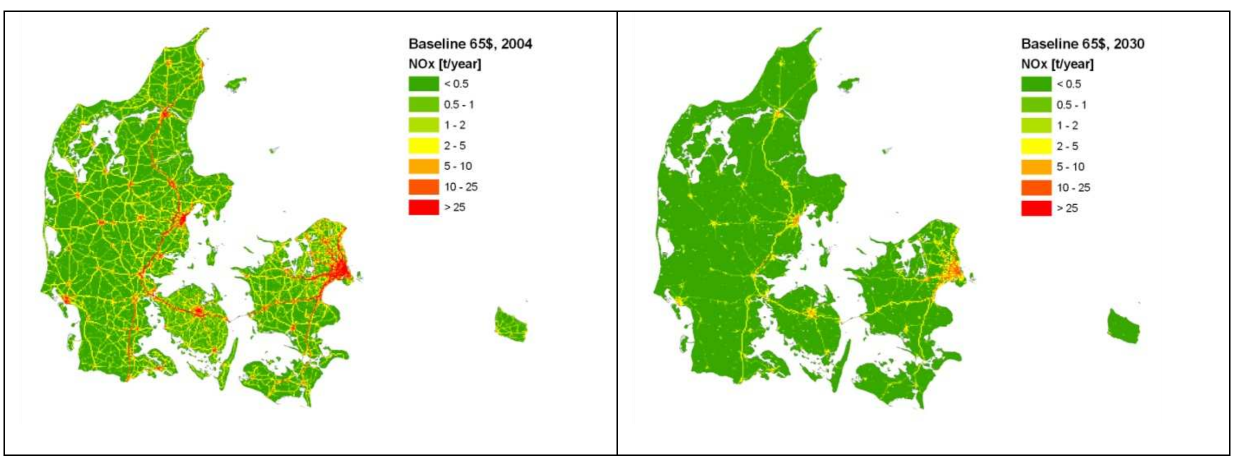

Figure 6: Baseline NO_x emissions for road transport in 2004 and 2030.

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Session 2: Energy System Modelling

Keynote speaker: Professor Henrik Lund, Aalborg University, Denmark

Chair: Allan Gross

Including Health Cost in the CEEH version of the Energy System Optimisation Model Balmorel

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Fossil fuel related air pollution influences both the natural environment and human health. The particle pollution from cars and trucks alone is considered to cause more deaths than traffic accidents. This has lead to the establishment of a Danish Centre for Energy, Environment and Health (CEEH) which is supported by the Program Commission for Energy and Environment under the Danish Council for Strategic Research. The objective of CEEH is to establish an interdisciplinary based system to support optimal future planning of energy production and usage with respect to costs related to the natural environment and human health. To ensure the needed interdisciplinary approach the centre includes researchers from meteorology, air pollution, environment, energy, physiology/health and economy. The main outcome of the centre is an integrated regional model system including components for air pollution chemistry and dispersion down to urban and sub-urban scales, and model components of the impacts on public health and the external environment.

A part of this integrated modelling system is the modelling of the energy system and how it should be configured the next 40 years when also taking costs of externalities into account in the economic optimisation of the future energy system in Denmark. This paper is a description of how the energy system model Balmorel is expanded to include more sectors and how externalities are integrated into the model.

The Balmorel model is a linear optimisation model of a power and heat system with perfect competition. Based on scenarios for the development in input parameters such as energy demand, fuel prices and technology data, the model calculates the operation of the units in the power system and the new investments in power plants and transmission lines that maximise social surplus in the power system. The model is multi-regional consisting of regions connected by transmission lines. It takes into account the balance between supply including net export and demand in each region, capacity restrictions for production units and transmission lines, technical restrictions for CHP plants, balance equations for heat, and hydropower. The externality costs have been build into the objective function in order to take these into account in the optimisation process.

Balmorel will be expanded to include transport, heating of buildings and industrial processes. The different modules are developed and have been developed by different researchers and are then integrated in one version of Balmorel. Developing some of the modules and the work of integrating the different parts is carried out in the light of CEEH.

Status of Balmorel features:

- **Electricity and heat sector ,storage etc. (Ravn 2001)**

Given power and heat demand, the model operates existing generation units and makes investments in new capacity in such a way that the resulting costs of the hole system are minimizid. Generation technologies available to the model are CHP, extraction, condensing and back-pressure power plants, heat only units, heat pipes, and various renewable technologies (i.e. hydro, wind, and solar power).

- **Heat savings module (Zvingilaite 2009)**

  This feature makes it possible to consider demand side measures to reduce heat demand. Given the stock of buildings, their potential for improvement and the associated cost the model has flexibility to choose whether to respond to heat demand simply by generating the necessary amount of heat or to invest in e.g. better insulating windows reducing the demand and then generating the rest, whatever is cheaper.

- **Transport (Meibom and Karlsson 2009)**

  So far fuels for transportation are produced in the model, but the demand for the different transport fuels is given exogenously. But we are working on implementing a detailed car choice model that also includes driving patterns for electric vehicles.

- **Hydrogen (Karlsson and Meibom 2008)**

  This part of the model enables utilisation of hydrogen-based technologies for electricity storage and transportation.

- **Industrial processes**

  This module is not yet developed.

- **Accounting for externalities (Brandt et al. 2009)**

  This part of the module allows taking into account the cost of environmental and health damage of an energy system. The cost is attributed to emission of a particular substance from technologies in the energy system, which influences both operation and investment decisions.

The different sectors using different kind of fuels in different geographical areas will each have different costs related to their air emissions due to population density in the area and the transport of the emissions in the atmosphere to other areas. The health impact from air emissions from the different sectors today will be illustrated and the impact of including health cost in the decision of future investments in the different sectors is discussed.

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Analysis of the impact of changing hydro-meteorological parameters on the electricity production of once-through cooled thermal power plants in Germany - A System Dynamics modelling approach

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Introduction

Most conventional thermal power plants in Germany use river water for cooling purposes. On the one hand, the temperature of the river water influences the performance of cooling systems and thus affects the production of electricity. On the other hand, the excess heat of power plants is discharged back into the river or into the atmosphere. The amount of heat discharge and thus temperature increase of the river water is constrained by legal threshold values that can force power plant operators to reduce the electricity output. A prominent example for such effects is the partial reduction of electricity production of several thermal power plants in Germany during summer 2006 (German Atomic Forum, 2007; Müller *et al.*, 2007). Since there is a high share of thermal power plant capacity in Germany and mean river water temperatures are expected to rise in the future, the overall electricity production can be considerably affected. This study is analysing these impacts by modelling selected cooling systems of thermal power plants based on a System Dynamics approach.

General description and modelling approach

This study is an interdisciplinary approach bringing together results of regional climate and water temperature modelling as well as energy system modelling based on the System Dynamics (SD) approach. System Dynamics is a methodical framework for the analysis of temporal and cross-linked interdependencies. The interactions in this study are reflected by the changing hydro-meteorological parameters in the context of climate change, the site-specific thresholds and the operation of the installed cooling system. The interdependencies of these components influence the intensity of impacts of climate change on the power plant output. As cooling system, a once-through cooling (OTC) was chosen and implemented in an SD model. The model was validated and tested for an actual power plant site. In the first instance, the analysis was carried out for one single power plant unit for which necessary data were readily available: the nuclear power plant (NPP) Kruemmel in Northern Germany. After model validation the impact of changing hydro-meteorological parameters like water and air temperatures on the cooling system was simulated and analysed.

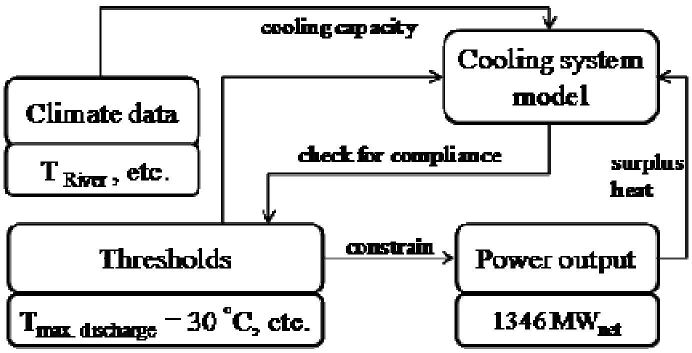

Figure 1: Schematic structure of the cooling system model with the most relevant components.

Model input data and data preparation

The input data include technical parameters, administrative regulations and hydro-meteorological parameters. Technical parameters of cooling systems are linked to thermodynamic processes. The following formulas and further thermodynamic relations (cf. Strauß, 2006 and Sauer, 1984) were integrated into the SD model:

The amount of heat Q_S (W) that is to be dissipated per time unit to the environment is specified in (1) (Sauer, 1984):

$$Q_S = Q_{Th} - Q_L - P_{el} = \frac{P_{el}}{\eta} - Q_L - P_{el} \qquad (1)$$

Where:

Q_{Th} = Thermal power (W)

Q_L = Heat loss to environment (W)

P_{el} = Net power output (W)

η = Efficiency (dimensionless)

The discharge temperature T_D (K) was calculated as follows:

$$T_D = \frac{Q_S}{W \cdot c_{H_2O}} + T_R + 273.15 \qquad (2)$$

Where:

T_R = River temperature (°C)

W = Cooling water amount (kg s^{-1})

c_{H2O} = Specific heat capacity of water (J kg^{-1} K^{-1})

River water related thresholds are set by the Directive 2006/44/EC of the European Parliament and of the Council. For the power plant site Kruemmel four threshold values considering river water and cooling water temperature apply:

1. temperature observed downstream of a point of thermal discharge must not exceed the unaffected temperature by more than 3 K (Directive 2006/44/EC),
2. thermal discharge must not cause the temperature downstream of the point of thermal discharge to exceed 28 °C (Directive 2006/44/EC),
3. temperature of the discharged cooling water must not exceed 30 °C (Vattenfall, 2010),
4. increase of cooling water temperature must not exceed 10 K (Vattenfall, 2010).

For model validation, observed time series of hydro-meteorological input parameters and power plant output data from 2006 were used. Air temperature data was taken from the climate station Boizenburg 18 km linear distance eastern and at Hamburg-Fuhlsbüttel 37 km linear distance north-west from the power plant site Kruemmel (German Weather Service DWD, 2008). Runoff data originates from the gauging station Neu Darchau around 36 km linear distance south-east of Kruemmel (German Federal Water and Shipping Administration WSD, 2009). Observed daily means of water temperatures were taken from the gauging station Schnackenburg situated about 90 km linear distance south-east of the power plant site (Project Group Elbe River Cleanliness ARGE Elbe, 2007). For the model validation we used power output data published in German Atomic Forum (2007).

For the analysis water temperature data for the future period (2011-2100) were estimated by using daily means of air temperature of the regional climate model REMO (UBA run) (Jacob *et al.,* 2001): A correlation analysis of homogenised observed water temperature series and daily air temperature series showed significant correlation between the two parameters. Therefore, water temperatures were simulated based on an exponential regression by using observed air temperatures as explanatory variable and thoroughly validated (Mohseni *et al.*, 1998; Mohseni & Stefan, 1999). Simulated air temperatures of REMO were validated for the control period 1961-1990 by using observed air temperatures of the DWD between 1961 and 1990. The seasonal cycle, monthly means and the standard deviation of observed and simulated air temperature data fit together well. Runoff was not modelled for the future. In this case average daily means of a 41-years period (1962-2002) and varied annual reduction in runoff were used to include possible future changes in hydrological parameters. To account for different climate scenarios, the analyses were carried out following three SRES (Special Report on Emission Scenarios) scenarios B1, A1B and A2 (Jacob *et al.*, 2008). Further site-specific parameters like cooling water quantity and efficiency factor were taken from Vattenfall (2010).

Model results

The model validation showed that the model is capable of reproducing load reductions of the power plant site Kruemmel as occurred in the past (e. g. summer 2006) (German Atomic Forum, 2007). The simulated mean annual deviations are 0.6 % and 2.3 % for the whole year and the summer months respectively. After validating the model, several scenario runs were carried out. Annual average load reductions from 2011-2100 were up to 3.4 %, 5.6 %, and 5.1 % for B1, A1B, and A2 scenario respectively (Table 1, see also Fig. 2 & Fig. 3), compared to the REMO Control Run 1961-1990 with reductions of up to 2.1 %.

	Annual average (%)				Average for June, July & August (%)			
	B1	A1B	A2	CR	B1	A1B	A2	CR
Mean	1.4	1.9	1.8	0.8	4.5	5.7	5.6	2.7
Max	3.4	5.6	5.1	2.1	11.7	15.4	16.0	7.2
Min	0.1	0.3	0.3	0.1	0.2	0.6	0.2	0.1

Table 1: Average relative reductions in power output for scenario runs (2011-2100) and REMO-control run (CR, 1961-1990) for whole years and the months June, July and August.

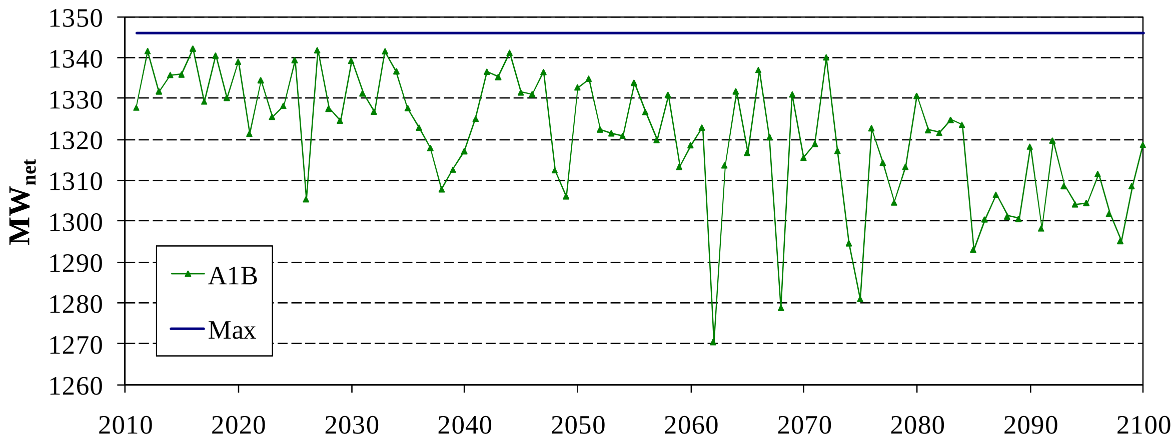

Figure 2: Average annual deviations from maximum power output (MW_{net}) for climate scenario A1B compared to maximum power output of the NPP Kruemmel.

The effect of varying river runoff on the cooling system was accounted for in sensitivity analyses. The variation of annual runoff reduction from 0.1 to 0.5 % per year (see Koehler, 2008) showed no increase in load reductions. The analyses revealed that for our model the reduction in power production is mainly caused by exceedance of the threshold for maximum cooling water temperature at discharge. In contrast, decreasing the other thresholds did not affect the model at first: For example, the threshold for maximum temperature increase of the river could be reduced by 50 % and the maximum river temperature threshold by 14 % until they affected the power output. The trends (Fig. 3) in decrease of power production are proven to be statistically significant using the Seasonal Mann-Kendall trend test ($\alpha = 0.01$).

In the model runs, reductions in power output only occurred in the summer months and were highest in July and August (Fig. 4). This is in accordance with the observed case in 2006 (German Atomic Forum, 2007).

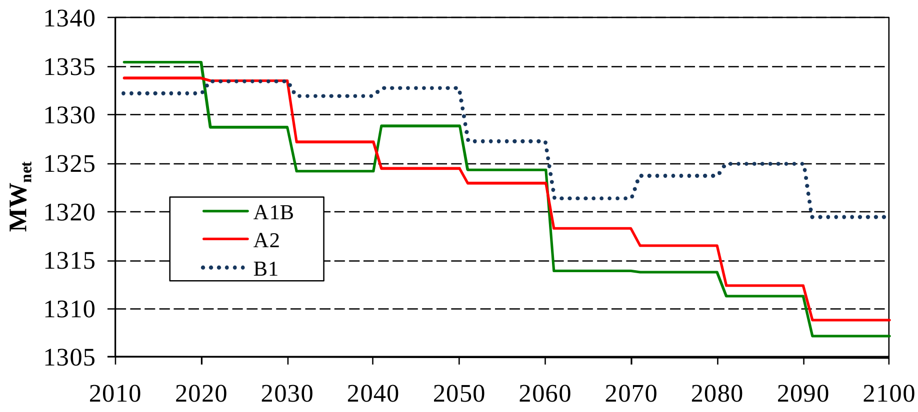

Figure 3: Decadal average deviations from maximum power output (MW_{net}) for three different climate scenarios A1B, A2 and B1. Negative trends confirmed by the Seasonal Mann-Kendall trend test.

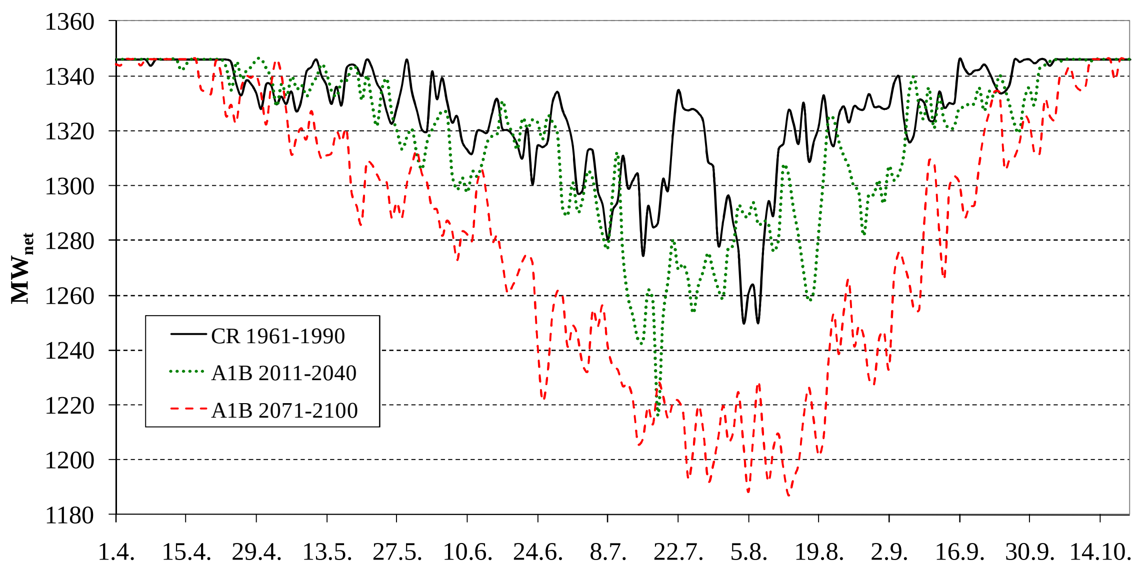

Figure 4: Simulated average deviations from maximum power output (MW_{net}) during months April to mid-October for daily 30-year means of scenario A1B compared to the Control Run of REMO.

Discussion and conclusions

The electricity production at the power plant site Kruemmel is likely to be affected by increasing air and thus water temperatures since output reductions already occurred in the past. When interpreting the results, it is important to consider that the SD model is only a simplified implementation of the cooling system with a daily temporal resolution. Some important aspects like electricity demand, maintenance, and losses in efficiency are not considered. However, the results of the validation reveal that the SD model is capable of reproducing the output reductions in the past and it can be considered an appropriate approach for the analysis of probable future reductions. Taking REMO model results as a basis for the simulation of the water temperature, these reductions account for average annual losses of 1.4 - 1.9 % and 4.5 - 5.7 % for the whole year and the months June, July and August respectively.

However, REMO only simulates the average development of air temperatures and cannot reproduce extreme weather conditions like the summers of 2003 and 2006. In combination with the possible rise of water temperatures, these extreme events may substantially aggravate reductions in power production (Förster & Lilliestam, 2009). Runoff reductions in the model did not affect power production at Kruemmel and probably might not present a problem at this site in the future.

Depending on the availability of site specific information the presented model can be adapted to other once-through cooled power plants. The results of the different climate scenarios can be used as an indicator for possible future losses in electricity production of individual power plants. These losses can be minimized in adaptation scenarios with regard to optimisation of power plant maintenance or possible retrofit of cooling systems. Therefore, the quantification of losses can support economic decisions with regard to possible investments.

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Modelling the demand for energy in an end-use perspective focusing on identifying energy efficiency opportunities

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Introduction

Energy balances or even energy chain representing the energy flows from primary energy to the demand are an important monitoring tool and are published by many national and international bodies, e.g., the IEA (IEA, 2009). In most of these statistics, the demand is generally characterized by broad economical sectors, such as industry, transportation, residential buildings and services. The balances show the quantity of each energy carrier that arrives to each of the sectors but do not show for which purposes they are used. While this highly aggregated approach is very useful for observing trends at the sector's energy use over time, it does not show where energy is effectively being used into services (e.g., water heaters, pumps and lighting systems) or where energy efficiency gains could be obtained. Therefore, a more disaggregated approach with a breakdown to end-uses and equipments could, besides suggesting measures to improve efficiency due to technological upgrade, also help finding measures regarding shifting from one energy carrier to other that could provide the same service without losing quality and bringing some benefits as less primary energy use, less CO_2 emissions or less energy dependence (e.g., shifting from electricity to gas or even to solar energy though solar thermal collectors).

This work proposes a characterization to guide the construction of energy demand models considering end-uses at the domestic, services, industry and transports sectors.

Mapping the energy system

The challenge on mapping an energy system is to find a structure of breakdown of the energy services within each main activity sector that could be modelled in a way that would assist identifying and quantifying energy efficiency measures, taking into consideration the level of existing data and keeping the number of end-uses and technologies manageable.

Several authors base their breakdown of the energy system on previous studies or statistical data due to reliability of the information. Sometimes the level of desegregation from several models is not specific enough to estimate savings or even identifying the real efficiency measures. Here it is developed a characterization relying on both statistic data and sound research studies aiming at improving the current models found to guide energy efficiency measures and justifying gathering more.

The United States Department of Energy (DoE) developed an Energy Footprints map to show the flow of energy supply, demand, and losses in U.S. manufacturing industries with the objective of identify the sources and end-uses of energy helping to pinpoint the most energy intensive areas, giving guidance to saving opportunities and providing a baseline to estimate the benefits of improving energy efficiency (U.S. Department of Energy, 2009). Also, the Energy Information Administration (EIA) (US Energy Information Administration, 2009) tries to provide data about the major sectors of the economy in an extremely disaggregated manner, showing the main conversion technologies (from energy to service) divided by energy carrier and end-use. Those data were used as a reference to build a map of the energy system, where opportunities to save energy can be shown and energy efficiency measures can be estimated.

The proposed demand model can be divided and described by: 1) the sectors from the economy where the end-uses belong; 2) the end-uses, representing the main services needed by the society (e.g., ambient heating and cooling); 3) the useful energy need required by the end-use, being an attempt to show how far the actual energy uses are from the real energy needs; 4) the competing end-use technologies inside each end-use (e.g., gasoline car and methanol car); 5) the efficiency of each technology or the reference energy use if efficiency is not applicable or more difficult to characterize (e.g., annual energy use from fridges or energy use per passenger-kilometre); and 6) the activity level, as the ownership rate or the share from the total market size where the need is inserted, in order to aggregate and calibrate the energy used.

The domestic sector

Energy use in the domestic or residential sector is generally defined as the energy used by households excluding their transportation outside the residential boundaries. That use in the domestic sector can be opened according to needs from the household, as domestic hot water, ambient heating and cooling, lighting, cooking, entertainment, etc. The sizes of a household (physical size and number of members) are key factors in the proposed model due to its relation to energy use. In general, larger residences require more energy to provide basic needs as heating and cooling (more space to be heated or cooled and more heat transfer with the outdoor environment), and the higher the number of members, the higher the needs for comfort as hot water or laundry services.

The end-uses that most characterize the domestic sector according to (Schulz, 2007),(Fawcett, Lane, & Boardman, 2000) and (McNeil, Letschert, & de la Rue du Can, 2008) are: domestic hot water, ambient heating, ambient cooling, refrigeration, freezing, cloth washing, cloth drying, dish washing, multimedia, computers, lighting, cooking, lifting and others. Each end-use was studied here in order to find the most common technologies in the market that provide such services. As a result, there were found 47 technologies that attend all end-uses, working with several energy carriers, such as natural gas, biomass, electricity and solar energy.

Services

The services sector, also referred to as the commerce sector or the tertiary sector consists of businesses, institutions, and organizations that provide services and encompasses many different types of buildings and a wide range of activities. Those different types of buildings can be associated to the type of service that they provide, which despite the different type of building have some singular type of energy needs and, in especial, different distribution of the energy needs. Commonly, energy used for services such as traffic and public lights and city water distribution and treatment, are also categorized as services energy use.

Economic trends and population growth drive the services sector activities and its resulting energy use. The need for services (health, education, financial, and government) increases as populations increase. Economic growth also determines the increase in activities offered in the services sector. The gross domestic product (GDP) from the services sector, the floor area and the number of employees are the three main indicators of activity in the sector (Mairet & Decellas, 2009). Floor area is the most important indicator of activity in this sector because energy uses tend to be proportional to area even when GDP or employment fluctuates with economic cycles (IEA, 1997) and especially because of physical relations (e.g., heat demand is directly dependent on the size of a building). A frequent problem is that floor area is not generally measured in all countries.

In order to model the energy use one has to look into the services subsectors (e.g., health care, food sale, lodging and offices) and end-uses. Here what were defined were the end-uses, which despite the heterogeneity of the sector can be found, in higher or lower levels, in most of the subsectors. The end-uses that most characterize the services sector in the world were defined based on (IEA, 1997), (McNeil et al., 2008), (Bertoldi & Atanasiu, 2006) and (Gruber et al., 2008) as: Hot water, lighting, heating, cooling, driving motors, refrigeration, office equipments, computers, cooking and others. Each end-use was examined in order to point out the technologies available in the market that provide such services. As a result, there were found 55 technologies that attend all end-uses, working with several energy carriers, such as diesel, natural gas, biomass and electricity.

Industry

The most important and most energy relevant part of the industry sector is the manufacture of goods and products, which consists basically on three kinds of productions: raw materials (e.g., steel and pulp), intermediate goods (e.g., machines and engines) and final goods used by consumers (e.g., TVs and washing machines). As in the service sector, the energy use and the structure from the industry sector can be associated to physical output, physical process and monetary measures.

Energy use performance based on physical units is closer to a measure of the "technical efficiency" of an industry and hence can be linked more directly to technology performance. They can therefore be used to identify the potential for efficiency improvements through new technologies. Physical measures of output are useful when studying a particular product or process, but it is almost impossible to find a single material or product that could represent the whole industry. The use of monetary measures of value solves the aggregation problem by relying on a common unit of output specification. Therefore, this study proposes a characterization based on monetary units to measure the size, the structure and the growth from the industry sector, and the energy use and the technological status of the more general processes that are present in all manufacture industries. The processes and end-uses selected are based on studies developed by (U.S. Department of Energy, 2009), (Almeida, Fonseca, & Bertoldi, 2003) and (Ozalp & Hyman, 2006). They consist on conventional boiler use, process heating, process cooling and refrigeration, electric motor-driven processes, electro-chemical processes, facility HVAC, facility lighting, onsite transportation and others. There were found only 28 technologies attending all end-uses in all manufacturing industries. As for the other sectors, the technologies work with several energy carriers.

Transports

Energy use in the transportation sector includes the energy used in moving people and goods by road, rail, air, water, and pipelines. The road transport includes light-duty vehicles, such as automobiles, small trucks, and motorbikes, and heavy-duty vehicles, such as trucks used for moving freight and buses for passenger travel. Here, the transport sector is divided into two major groups as passenger travel and freight transport.

Energy use per passenger-kilometer is the most important indicator of energy intensity for comparing modes of transportation (IEA, 1997) for passenger travel. Vehicle (mode) utilization and load factor explain part of the mobility differences among modes and individual fuel efficiency explains differences among technologies using same fuel and among different fuels. The fleet's age and energy technologies of each type of transportation mode dictate the fuel efficiency in a region.

Passenger travel can be generally seen as individual road transportation, mass transportation by road, rail, water and air and non-motorized modes, as cycling and walking. Individual road transportation is characterized by the most used mean of transportation, the automobile as car. Cars include personal light trucks and small vans. The cars can be distinguished between them according to the fuel technology, represented here by 14 technologies: diesel, gasoline, hybrid diesel, hybrid gasoline, ethanol, liquefied petroleum gas (LPG), compressed natural gas (CNG), hydrogen, biodiesel, diesel fuel cell, gasoline fuel cell, methanol fuel cell, plug-in hybrid electric vehicle (PHEV) and battery electric vehicle (BEV). Mass transportation is also divided by fuel technology as buses using diesel, gasoline, hybrid diesel, methanol, LPG, CNG, hydrogen, diesel fuel cell and electric buses for road transportation, rail transportation by diesel and electricity, water transports by diesel and fuel oil and air transportation by jet fuel.

The structure of the freight subsector is made up by some elements as the stock of vehicles, the distance traveled, the characteristics of freight and its quantity, and the utilization, usually measured in tones-kilometers. Utilization is one of the main indicators composing the energy intensity, showing the weight carried and the distance moved, representing an equivalent of the mobility indicator for freights. The energy intensity also rely on the modal choice, the fuel choice and the fuel efficiency, which along with utilization, serve to explain and compare freight energy use among modes and over time. As for passengers' transportation, the fleet's age and energy technologies of each type of transportation mode dictate the fuel efficiency in a region.

Freight transport can be grouped as land freight, by road and rail, water freight, by rivers and sea, and air freight. In this work road freight is represented by diesel, gasoline, ethanol, LPG, CNG, hydrogen and electricity (BEV). Rail freight is characterized by diesel, coal and electricity. Freight by waterways is represented by diesel and fuel oil technologies. And air freight is represented by jet fuel.

A test for the demand model: Portugal 2006

The proposed demand model was applied to Portugal and calibrated with 2006 energy and statistical data, showing that a more detailed model focused on the technologies can be used.

The application of the model to Portugal enabled the identification of the most important end-uses in each of the four sectors covered by the model, the energy carriers responsible for the end-uses and the respective amount of energy used by each end-use and carrier. Figure 1 illustrates the energy use for the services sector. Besides the respective energy use for each of the 41 end-uses by energy carrier for the 4 sectors, the model enables a characterization from the end-uses by 177 technologies, where opportunities on energy efficiency improvements due to technological upgrade of equipments and infrastructure (e.g. increasing insulation on houses) and the shift among energy carriers and service providers (e.g. shifting from cars to public transport) can be explored. This approach not only allows the quantification of physical (technological) efficiency gains, but also allows comparing energy savings using different perspectives, as emissions comparisons through technologies and energy carriers, or prioritizing the use of a specific energy carrier.

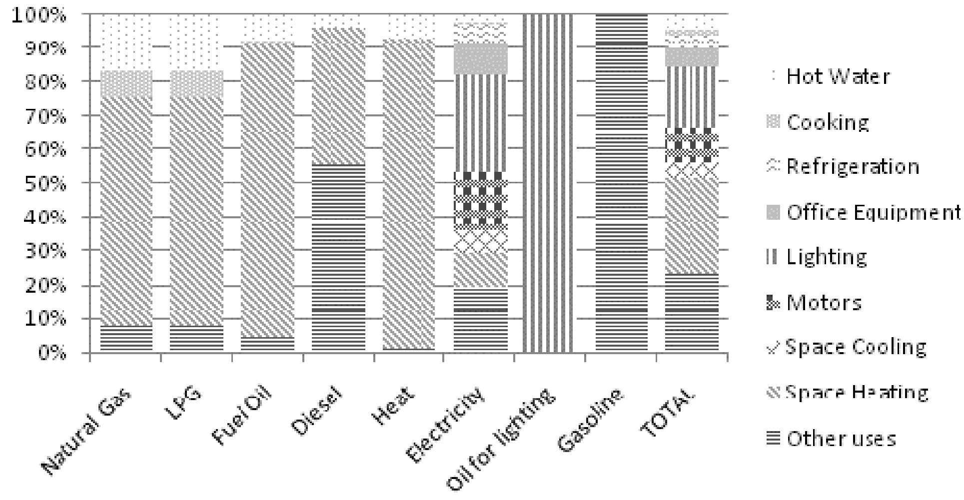

Figure 1: Share of final energy use by end-use and energy carrier

The proposed model opens a door to compare energy efficiency measures at the technological level with more quantitative detail. It also enables comparisons from the results from energy efficiency measures between end-uses and between sectors, showing where measures can be more effective.

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Bottom-up estimation of annual investment costs in end use technologies and their energy-using components

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Introduction: Energy Technology Investment Costs.

Investments in end use technologies are integral to energy system objectives including climate change mitigation. Across a wide range of future scenarios, reducing energy intensity by improving the efficiency of end use technologies provides cheaper and nearer-term alternatives to decarbonising the energy supply (Riahi et al. 2007; Ürge-Vorsatz & Metz 2009). Despite this broad acceptance of the pivotal role to be played by end use technologies in the coming decades, technologically-explicit energy system models resolve the demand-side poorly (Hanaoka et al. 2009). This influences modelling outcomes. A comparative review of 'bottom-up' or technologically-explicit energy system models with 'top-down' or equilibrium / econometric type macroeconomic models found that the former privileged supply-side decarbonisation to a greater extent: "*A likely explanation is that energy system models are relatively rich in technologies included in energy supply and thus see considerable options to reduce emissions*" (p5133, van Vuuren et al. 2009). Arguably, this emphasis on large-scale supply-side technologies, and the biases that may result, characterises not just energy system models, but also the policy community. With respect to end use technologies in buildings, the IPCC authors conclude: "*in the vast majority of countries, detailed end-use data is poorly collected or reported publicly, making analyses and policy recommendations insufficiently robust*." (p437, Metz et al. 2007).

One area in which this comparative dearth of detail of end use technologies is manifest relates to investment or diffusion costs. Each year, the International Energy Agency ('IEA') publishes an authoritative annual report, the World Energy Outlook, which provides a detailed evaluation of the dominant challenges for the global energy system as well as projections for its development over a decadal timeframe. The World Energy Outlook reports are rich in sectoral and technological assessments on both supply- and demand-sides, informed by the IEA's own data and models as well as peer-review. For investment costs, however, total investments requirements in the reference and other scenarios are presented only for supply-side technologies. The most recent World Energy Outlook identifies cumulative capital investment costs in the reference scenario of $26 trillion (in 2008$) in the period to 2030, or in the region of $1 trillion a year (p104, IEA 2009). *Total* investment costs in end use technologies, however, are not quantified. However, the IEA – like other modelling teams – do regularly quantify *incremental* investments required in selected end use technologies (usually more efficient ones).

In sum, end use technologies are not treated in the same way as supply-side technologies. Investment data and analysis are more aggregative, less quantitatively specific, and tend to be in incremental not total terms. This risks reinforcing the modelling bias towards large-scale supply-side investments.

Research Method & Issues

We present a global, bottom-up estimate of total annual investment costs in end use technologies. We used volume data (production, delivery, sales, installations) and cost estimates to approximate total investment costs in 2005 in both end use technologies and their specific energy-using components (see below for explanation). We included low and high sensitivities around central estimates, taking account of uncertainties in both volume and cost assumptions. Our intention is to provide a first order point of comparison with the $1 trillion per year supply-side investment requirements cited by the IEA.

To ensure comparability between supply-side and demand-side investments, a common definition of the unit of analysis is needed. Supply-side investments are quantified at the level of the power plant, refinery or LNG terminal. These are each complex, integrated technological systems with an energy conversion technology at their core (the steam turbine, the distillation and cracking unit, the liquefaction train). These 'energy-converting components' are configured within their corresponding technological system to provide a useful service to intermediate users (utilities, fuel distributors, pipeline or shipping companies).

The logical demand-side analogues of these technological systems are the aircraft, vehicle, fridge and home heating system. Although generally less complex, each of these technological systems similarly has an energy conversion technology at their core (the jet engine, internal combustion engine, compressor, boiler). In addition, each is configured to provide a useful service to final users (air passengers, commuters, households). With demand-side technologies, however, this definition of the unit of analysis is problematic. Investments in (and performance of) end use technologies are dependent on investments in associated infrastructure such as airports, roads and buildings. Is it meaningful to quantify the investment cost of a home heating system without quantifying the investment cost of a home? Is the end use technology a boiler or a building?

Although the same issue exists on the supply-side, it is largely addressed by additionally quantifying investment costs in associated transmission and distribution infrastructure. The problem on the demand-side is that the same approach would result in a sum of the total investment costs in all building structures, roads, railways, ports, airports, industrial plants, equipment, appliances, and so on ... The IEA recognise this *reductio ad absurdum* problem, but their response is to use incremental rather than total investment costs for end use technologies whose unit of analysis is not clear. The result is a skewed picture of investment needs and priorities which mixes together both total and incremental costs, and systemic and component technologies.

Here, we argue that the arbitrariness of the definitional boundary should be as consistent as possible. We quantified two sets of end use technology investments costs. Our first, broader definition and data set describe *end use technologies* as the smallest (or cheapest) discrete purchasable units by final consumers. This implies boilers and air conditioning units not houses, and dish washers and ovens not kitchens. Our second, narrower definition and data set describe the specific *energy-using components of these end use technologies*. This implies engines in cars, and light bulbs in lighting systems. Table 1 summarises these distinctions for the technologies analysed. In some cases (industrial motors, mobile heating appliances), a distinct energy-using component was not identified and so the data in both cases are the same.

End Use Service	End Use Technology	Energy-Using Component
mobility	commercial jet *aircraft*	jet *engine*
mobility	*vehicles* (cars and commercial)	internal combustion *engine*
space conditioning	*central heating systems* (boiler/furnace, ducts/pipes, radiators, controls, & network connections for new systems)	*boiler* or furnace
space conditioning	*air conditioning systems* (AC unit, ducts, controls, & network connections for new systems)	*air conditioning unit*
space conditioning	*mobile heating appliances* (e.g., portable convection / fan heaters)	- (same as for technology)
lighting	*lighting* (light bulb + fixture)	*light bulb* (or lamp)
food storage, cooking, cleaning	large *household appliances* (fridges, freezers, clothes washers & dryers, dish washers, cookers)	*compressors, motors, fans, heating elements* (depending on appliance)
various (e.g., processing)	*industrial motors*	- (same as for technology)

Table 1. Summary of Technologies & Components Included in Calculations.

Research Findings

We estimate that investment in 2005 in end use technologies was in the order of \$1 - 3.5 trillion; we estimate that investment in 2005 in the energy-using components of these end use technologies was in the order of \$0.1 – 0.7 trillion. The breakdowns of these totals by technology are given in Figures 1a & 1b.

We emphasize that these investment cost ranges are under-estimates. Although we aimed to capture the principal end use technologies in terms of the costs of their energy-using components (not the technologies themselves), investment costs in many technologies were not quantified. These include: all propeller-based and non-commercial aircraft, helicopters, all military technologies, mass transit systems, water heaters (residential and other), information and communication technologies, small appliances, other consumer electronics, and all industrial equipment and process other than motors (e.g., blast furnaces, pulp mills, cement kilns).

With the exception of industrial plant, we believe the inclusion of these categories would not substantially increase the investment cost range for energy-using components; however, they would substantially increase the investment cost range for end use technologies.

End Use Technologies in 2005		low sensitivity	central estimate	high sensitivity
GRAND TOTAL COSTS	**$bn**	**984**	**1,739**	**3,549**
commercial jet aircraft	$bn	*12*	***28***	*50*
cars	$bn	*540*	***758***	*1,194*
commercial vehicles	$bn	*270*	***427***	*672*
buildings (retrofits) - central heating systems	$bn	*47*	***250***	*979*
buildings (new) - central heating systems	$bn	*33*	***93***	*248*
mobile heating systems	$bn	*2*	***4***	*5*
buildings (retrofit) - air conditioning systems	$bn	*9*	***42***	*137*
buildings (new) - air conditioning systems	$bn	*7*	***20***	*41*
lighting	$bn	*17*	***38***	*83*
large household appliances	$bn	*45*	***75***	*124*
industrial motors	$bn	*2*	***6***	*16*

Figure 1a. Estimated Investment Costs In Selected End Use Technologies.

'Energy-Using Components' of End Use Technologies in 2005		low sensitivity	central estimate	high sensitivity
GRAND TOTAL COSTS	**$bn**	**124**	**297**	**713**
commercial aircraft - jet engines	$bn	*3*	***7***	*13*
cars - engines	$bn	*36*	***76***	*159*
commercial vehicles - engines	$bn	*27*	***57***	*119*
buildings (retrofits) - central heating units	$bn	*13*	***52***	*158*
buildings (new) - central heating units	$bn	*9*	***20***	*41*
mobile heating units	$bn	*2*	***4***	*5*
buildings (retrofits) - air conditioning units	$bn	*5*	***21***	*69*
buildings (new) - air conditioning units	$bn	*4*	***10***	*20*
lighting	$bn	*12*	***27***	*59*
large household appliances	$bn	*11*	***18***	*53*
industrial motors	$bn	*2*	***6***	*16*

Figure 1b. Estimated Investment Costs In 'Energy-Using Components' Of Selected End Use Technologies.

Given the definitional problem described above, we argue that the appropriate point of comparison for estimates of supply-side investment costs is a range spanning the narrow category of 'energy-using components' at the lower end to the broader category of 'end use technologies' at the upper end (see Table 1 for details).

Taking into account the extent of end use technologies missing from this analysis, we argue that the range of demand-side investment costs is conservatively in the order of $0.3 – 4.0 trillion each year. This compares with estimates of annual supply-side investment costs in the order of $1.0 trillion. Note also that the demand-side investment cost data is for 2005 and so excludes 'forcing' of investments through climate policy or otherwise. Although the two ranges span the same orders of magnitude, the upper bound of demand-side investment costs is 4 times higher than its supply-side equivalent, noting also that this is likely a (potentially substantial) under-estimate. Interestingly, this aligns with the IEA's estimation that demand-side investment needs exceed supply-side investment needs by a factor of 4 - 5 (IEA 2008).

Disaggregating the data by technology shows clearly the dominance of transportation (see Figure 2). Disaggregating the data by region shows that approximately two thirds of the end use investments costs in 2005 are in OECD countries and the former Soviet Union; the remaining one third are in developing economies.

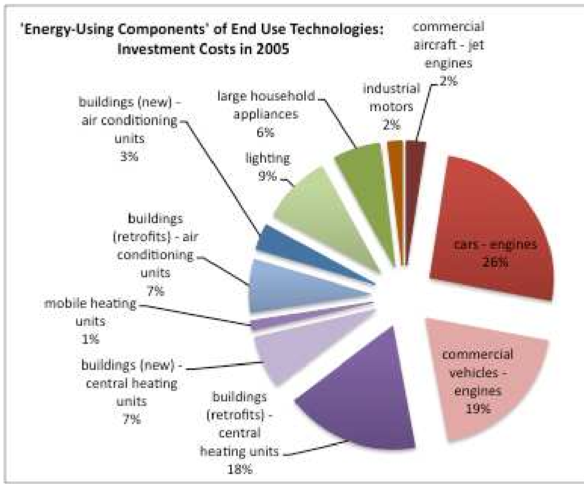

Figure 2. Proportional Breakdown Of Estimated Investment Costs In 'Energy-Using Components' Of Selected End Use Technologies.

Implications of Findings

The magnitude and diversity of the end use technology investment cost data reinforces the importance of redressing the supply-side bias of energy system analysis. This has implications for both the modelling and policy communities:

1. Energy system models need to better resolve end use technologies, and better represent diffusion processes and adoption behaviors;

2. Public R&D and investment portfolios need greater balance between large-scale supply-side and granular end use technologies;

3. Greater attention should be placed on developing and testing end use technology innovations (including adoption behaviors) and scaling them from demonstration projects to mass market.

4. Efficiency and other performance standards (and their enforcement) must play a critical role in driving demand-side changes;

5. End user fragmentation and dispersion emphasizes the importance of market transformation activities focused on more highly concentrated points of the end use technology supply chains.

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GIS based Model to optimize the utilization of renewable energy carriers and related energy flows

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A significant part of final energy consumption is demand for space heating. This demand is mainly met by fossil fuels. Given the challenge of climate change and question marks over energy security and import dependency on fossil fuels, improvements in energy efficiency and greater use of renewable energy may be important policy considerations.

The paper presents a modelling approach for the optimization of the fulfillment of the heating demand within a defined region of interest, favoring renewable energy carriers - with a particular focus on spatial differentiation. The modelling approach that is presented handles information on geographically disaggregated data describing renewable energy potentials (biomass, solar energy, geothermal energy, ambient heat) on the one hand and geographically disaggregated information on the heating demand on the other hand. This spatial balance is the basis for modelling an optimum spatial utilization of identified renewable energy resources to satisfy the heating demand with respect to the objective function of the model, which is defined as highest economic efficiency with respect to greenhouse gas emissions constraints in the region. To take into account the spatial relevance of the single elements of the energy system in an appropriate way, all relevant spatial data are disaggregated to a consistent spatial resolution. This includes the energy potentials, the demand structure as well as some infrastructure data. The region of interest is segmented into a collection of raster cells, which present the smallest spatial unit in the model. The smallest size of raster cells is 250 m x 250 m.

The general model framework within this approach consists of three parts:

- **The potential model** – includes separate models to estimate the potential of individual renewable energy carriers (biomass, solar energy, ambient heat) in a spatially disaggregated way with their specific characteristics. These separate models are integrated into the overall potential model.
- **The demand model** – illustrates the spatially disaggregated heating demand, expressed as heating degree days. This is the basis for the estimation of the effective demand in relation to the insulation standard of buildings.
- **The dynamic fulfilment model** – is used to derive an optimized setup of the energy system for the fulfillment of the heating demand. For the generation of various scenarios a distinction is made between fixed parameters defined by the system (present situation) and variable parameters (e.g. future costs). The variable parameters (insulation standards, domestic fuel type, natural gas and district heating grid, fuelling of power plants and use of renewable energies) are defined differently for the development of the scenarios. Depending on these definitions, a sensitivity analysis can be carried out. The model is implemented as a linear optimization model realized in the modelling language GAMS.

The use of scenario analysis allows the testing for key sensitivities in the model. These outcomes may have important policy implications or provide strategic information to stakeholders.

Session 3: Economic Valuation

Keynote speaker: Professor Susan Chilton, Newcastle University, United Kingdom

Chair: Kenneth Karlsson

Problems of discounting and aggregation in estimating the value of a life year *VOLY* on the basis of persons willingness to pay *WTP*

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Introduction

Valuation of changes in risk of death is often based on the value of a life year *VOLY* - cf. Nielsen (2008). If a change in death risk - e.g. as a result of a change in air quality - is described as a change Δs_j^t in person *j*'s age-dependent survival probabilities s_j^t it is easy to calculate how many life years $\Delta LY(t)$ are lost per year *t* or in total $\Delta LY(N)$ for the existing population of *N* persons.

$$\Delta LY(t) = \sum_{j=1}^{N} \Delta s_j^t \qquad \Delta LY(N) = \sum_{j=1}^{N} \sum_{t=0}^{\infty} \Delta s_j^t$$

The loss of life years can subsequently be multiplied with an estimated *VOLY* to get the total value of the change in death risk - loss of welfare - for all existing persons in one year *t* or over their expected remaining life time.

The purpose of welfare economic valuation is to calculate an indicator of changes in persons expected life time utility. Therefore it is important that *VOLY* is estimated in such a way that it can be used for this purpose. When estimation is based on results of contingent valuation studies it should be possible to interpret every single person's one-off willingness to pay $WTP_j(1\ year)$ for a life year as an indicator of his loss in life time utility. Economists normally assume that changes in expected life time utility ΔLU_j^e depend on changes in expected life time consumption ΔLC_j^e and that this change can be calculated as the present value of expected consumption changes in every future year from *t = 0* when the change in death risk is assumed to take place to *t =* L_j^e where the person is expected to die. I.e.

$$WTP_j(1\ year) = \Delta LC_j^e = \sum_{t=0}^{L_j^e} \frac{\Delta s_j^t \cdot C_j^t}{(1+i_j)^t}$$

Here $\Delta s_j^t \cdot C_j^t$ is the change in expected consumption for person *j* in year *t* and i_j is person *j*'s individual discount rate.

If this assumption is accepted the aggregation of persons $WTP_j(1\ year)$ for a life year to get *VOLY* is complicated by three problems:

- Persons' $WTP_j(1\ year)$ reflect individual time preferences - i.e. individual discount rates i_j - and in cost benefit analyses the value of loss of life years in each year is discounted with the social discount rate *i*.
- The $WTP_j(1\ year)$ of persons with different age reflect their different expected rest life time L_j^e - i.e. $\sum_{t=0}^{L_j^e} \Delta s_j^t = 1$ for all persons *j* - the time horizons of their expected consumption losses are different.
- If persons of different ages all are asked about their willingness to pay for one life year they are in fact asked about different changes in death risk.

These three problems mean that *VOLY* cannot be calculated as a simple mean of persons' *WTP*. It is necessary to correct the *WTP* for the differences in discount rates and time horizons so that the aggregation of the individuals' *WTP* and the calculation of *VOLY* are based on corrected *WTP*. However, the correction and calculation of *VOLY* can be made in different ways and with different results. This can be illustrated by simple calculations. It is not clear which correction and aggregation method is the most correct.

Different ways of calculating *VOLY* based on persons' *WTP* for a life year

The simplest way to calculate *VOLY* is to calculate it as the mean of persons' one-off willingness to pay for one extra life year - i.e.

$$VOLY = \frac{\sum_{j=1}^{N} WTP_j(1\ year)}{N}$$

This method is unsatisfactory because it does not take into account that the interviewed persons have different individual discount rates, have different ages and expected rest life time and therefore have expressed willingness to pay for different changes in their age dependent survival probabilities. You can correct for these differences in different way.

1.

You can calculate a corrected willingness to pay for each person based on his individual discount rate, expected rest life time and the social discount rate - i.e.

$$WTP_j^{corr1}(1\ year) = \sum_{t=0}^{L_j^e} \frac{WTP_j(1\ year) \cdot a(i_j, L_j^e)}{(1+i)^t} \quad \text{and} \quad VOLY_1 = \frac{\sum_{j=1}^{N} WTP_j^{corr1}(1\ year)}{N}$$

where $a(i_j, L_j^e)$ is the capital recovery factor for a discount rate i_j and a time horizon L_j^e. In this way you take into account the persons' different individual discount rate and make their calculated yearly *WTP* comparable by discounting them with the social discount rate *i*. But, the fact that the persons different WTP_j reflect different expected rest life times is not taken into account when calculating $VOLY_1$ as the simple mean of the corrected $WTP_j^{corr1}(1\ year)$.

2.

One possible solution to this problem is to calculate *VOLY* as the present value of the mean of the persons' calculated yearly willingness to pay. In calculation of the present value a common mean expected rest life time L_{mean}^e is used. In this way $VOLY_2$ is calculated in the following way.

$$WTP_j(yearly) = WTP_j(1\ year) \cdot a(i_j, L_j^e) \quad \text{and} \quad WTP_{mean}(yearly) = \frac{\sum_{j=1}^{N} WTP_j(yearly)}{N}$$

$$VOLY_2 = \sum_{t=0}^{L_{mean}^e} \frac{WTP_{mean}(yearly)}{(1+i)^t}$$

Perhaps this is not a satisfactory solution, because on the one hand the persons' different time horizons L_j^e are respected in calculating their yearly willingness to pay $WTP_j(yearly)$ and on the other hand a common mean expected rest life time L_{mean}^e is used to calculate $VOLY_2$ as the present value of the mean yearly willingness to pay $WTP_{mean}(yearly)$.

It might be better to assume the same mean time horizon in calculating both the yearly willingness to pay and *VOLY* as the corrected present value. This is of course a very strong assumption, but it solves the problem of aggregating WTP_j expressed for different time horizons. Moreover, empirical experiences show that persons are not always very aware of the time horizon over which they express their willingness to pay.

3.

If the assumption of a common mean time horizon L^e_{mean} is accepted, then a corrected $VOLY_3$ can be calculated in the following way.

$$WTP_j^{corr3}(1\ year) = \sum_{t=0}^{L^e_{mean}} \frac{WTP_j(1\ year) \cdot a(i_j, L^e_{mean})}{(1+i)^t} \quad \text{and} \quad VOLY_3 = \frac{\sum_{j=1}^{N} WTP_j^{corr3}(1\ year)}{N}$$

Compared to the two previous calculations 1. and 2. this calculation has the advantage that information about every single person's time horizon L^e_j is not needed. However information about his personal discount rate i_j is still needed. This is a weakness because it might be difficult in practice to get this information - in fact even more difficult than to get information about the personal time horizon.

Again a possible solution might be to assume a common individual discount rate i_c. Empirical analyses have shown that this rate can be very high compared to the social discount rate *i* - cf. Cropper et. al. (1992) and Lau (2001). There is even indication that persons' time preferences can be best described by hyperbolic discounting, but in the following this is ignored.

4.

If a common individual discount rate i_c is assumed $VOLY_4$ can be calculated in this way.

$$WTP_j^{corr4}(1\ year) = \sum_{t=0}^{L^e_{mean}} \frac{WTP_j(1\ year) \cdot a(i_c, L^e_{mean})}{(1+i)^t} \quad \text{and} \quad VOLY_4 = \frac{\sum_{j=1}^{N} WTP_j^{corr4}(1\ year)}{N}$$

Of course this calculation will lead to another but not necessarily a better result than each of the previous calculations 1. - 3. In fact it is unclear which calculation of *VOLY* should be preferred. It is not easy to aggregate persons' WTP_j into one *VOLY* the when the WTP_j reflect different individual discount rates and time horizons and it shall be possible to use the resulting *VOLY* in cost benefit analyses where a social discount rate *i* is used.

Perhaps the idea of calculating one representative *VOLY* on the basis of persons' expressed willingness for a life year should be abandoned. Perhaps we should not concentrate on total life years lost in our description of changes in death risk which is the reason why estimating *VOLY* becomes so important. It might be more consistent with the general methodology of cost benefit analysis to concentrate on changes in persons' age dependent survival probabilities and the change in the persons' expected consumption in each year that this gives rise to.

Valuation of changes in death risk based on changes in persons expected consumption over their life time

From a person's one-off willingness to pay $WTP_j(\Delta L_j^e)$ for a certain change in his expected rest life time $\Delta L_j^e = \sum_{t=0}^{\infty} \Delta s_j^t$ the value of his yearly consumption C_j if he is alive can be calculated.

$$WTP_j(\Delta L_j^e) = \Delta LC_j^e = \sum_{t=0}^{\infty} \frac{\Delta s_j^t \cdot C_j}{(1+i_j)^t}$$

This assumes that the $WTP_j(\Delta L_j^e)$ reflects the present value of the change in expected life time consumption ΔLC_j^e and that we know persons individual discount rates i_j and the changes in their age-dependent survival probabilities.

From the value of each person's yearly consumption C_j it is possible for each year to calculate the loss of expected consumption $\Delta s_j^t \cdot C_j$ for all possible changes in the survival probabilities. For each year *t* these losses of expected consumption can be summed up for the persons that are alive in this year N_t to calculate the total loss of expected consumption ΔC_t^e for that year. We have

$$\Delta C_t^e = \sum_{j=1}^{N_t} \Delta s_j^t \cdot C_j$$

This calculation is consistent with general cost benefit methodology. In traditional *CBA* it is the projects consequences for persons' expected consumption (in a wider sense including i.e. environmental goods) which are described and valuated in each year over the chosen time horizon. A change in death risk will also affect persons' expected consumption in each year - however not by changing the level of consumption if the persons are alive, but by changing their chances of being able to consume.

As in traditional *CBA* the present value of a change in death risk $NPV(\Delta s)$ can be calculated by discounting the total change in expected consumption in each year ΔC_t^e with the social discount rate *i*.

$$NPV(\Delta s) = \sum_{t=0}^{\infty} \frac{\Delta C_t^e}{(1+i)^t}$$

When persons die their change in expected consumption does not count any longer. It is replaced by the change in expected consumption for future persons, which can be assumed to correspond to the change for young living persons. The expected change in their age dependent survival probabilities and their expected level of consumption can be assumed to be the same if no other information is available.

In this way we respect that persons lose life years over many years and that persons are affected differently by a change in the risk of death. We can describe how the expected life time consumption and utility of old people, younger people and future people are affected differently by a change in the risk of death. Hereby it becomes possible to take the distribution aspect into account, which normally is not done when the valuation is based on *VOLY*.

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From the Value of a Statistical life to the Value of a Life Year lost. Does it make sense?

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Introduction

Within the traffic sector the preference-based Value of a Statistical Life (VSL) has emerged as the standard approach to valuing changes in mortality risk. The VSL reflects the population mean of the marginal rate of substitution between wealth and risk (Jones-Lee 1991) and is computed by firstly estimating the mean willingness-to-pay (WTP) for a small mortality risk reduction and then dividing by the risk reduction in question.

In the evaluation of healthcare services the main approaches applied have been the cost-effectiveness (CEA) and cost-utility analyses (CUA), in which case the benefits are measure in life years lost or Quality Adjusted Life Years lost (QALYs). It has been suggested that a predetermined VSL from the traffic sector could be used to derive a WTP- based Value of a Life Year (VOLY) (Mason et al., 2008; Hirth al., 2000) which could then be used as a WTP threshold in conjunction with CEA and CUA. A number of important issues related to the task of eliciting and applying a WTP threshold have been raised in the literature (see Gyrd-Hansen, 2005 for an overview). However, this paper will focus on the appropriateness of applying a VSL to disclose a VOLY by focusing on the validity of the VSL-VOLY relationship.

One criterion for using a VSL from the traffic sector as a starting point in the health care sector is the validity of benefit transfer across contexts. There is substantial evidence that people perceive certain hazards as worse than others and that some deaths seem worse than others (Sunstein 1997). Hence, it can be questioned whether individuals' preferences for mortality risk reductions are independent of the context (i.e. health and traffic) in which they have been derived. Policy applications involving the WTP-based VSL tend to employ a constant VSL across age groups (Baker et al., 2008) even though VSL in numerous empirical surveys have been shown to vary with age (Jones-Lee et al., 1985; Kidholm 1995). Applying a constant VOLY across age groups would be desirable from a policy perspective since then there would be no need for discriminating between different age groups. However, employing a constant VOLY implicitly requires that VSL decreases with age which appears to be a potential contradiction to the standard approach of applying a constant VSL. It is possible that a constant VOLY across age groups will produce wildly inaccurate measure of benefits, since the variability of VOLY with age could be very high (Sunstein 2004).

The aim of this paper is twofold. Firstly, based on the empirical survey, we investigate whether the VSL in the present study is constant across context. Secondly, applying the existing formulas for calculating a VOLY indirectly from a VSL we analyze whether a constant VOLY can be derived from the present data set.

Background

With the theoretical starting point in expected utility theory, the one-period VSL approach has been very well developed theoretically (Baker et al., 2008; Jones-Lee 1989) and the empirical estimation techniques are advanced (Carthy et al., 1999;Braathen et al., (2009). However, as emphasised in (Moore and Viscusi 1988); "Analysts have long noted that the appropriate value of life for policy analysis cannot be divorced from the duration of life involved since lives are extended, not permanently saved" (Moore and Viscusi 1988, quote p. 370). For example, many environmental policies involve actions today whose effects on the probability of dying are realised at some time in the future. When investigating the probability of death in a multi-period model "number of premature deaths" (or prevented fatalities) is according to (Rabl 2003) not a meaningful indicator since sooner or later everybody will die and a decrease in the "daily death count" will always be followed by a later increase in the "daily death count". Instead, loss of life expectancy is the appropriate indicator. The monetary benefit from policies can vary a good deal according to whether prevented fatalities or life-years saved are counted and much debate has surrounded the question of whether the VOLY or the VSL framework should be implemented in cost-benefit analyses. The US Office of Management recommends that cost-benefit analyses undertaken in their agencies apply both the VOLY and the VSL approach (Sunstein 2004), whereas the OECD guideline proposes the use of a mix[1] of VSL and VOLY for cost-benefit analyses in the context of air pollution (Pearce et al., 2006).

Does context matter?

Individuals' preferences for a reduction in the risk of dying prematurely by some particular cause may – in addition to the magnitude of the risk reduction *per se* – depend on a variety of context-related factors. In this paper, "context-related factors" will be used as the generic term for all risk characteristics with the exception of the magnitude of the probability of death *per se.* There is a large literature on how context-related factors influence WTP and the following risk characteristics have been identified; voluntariness of exposure, controllability, dread, severity, private knowledge, public knowledge, and private exposure (McDaniels et al. 1992). In addition, Viscusi (1979) argues that the personal baseline risk for the cause of death concerned might influence the individual's risk preferences over alternative options. Chilton et al., (2006) demonstrate that it might be rational for an individual to prefer a change in the risk for a cause with higher baseline

[1] More specifically, VSL for an 'acute death' (based on results from time-series studies in which it is estimated how daily air pollution levels affect the number of deaths each day) and VOLY for 'chronic death' (based on cohort studies on people living in different cities).

risk against a risk change for a cause with relatively lower baseline risk. The validity of benefit transfer across different contexts will depend on whether individual's preferences for mortality risk reductions are independent of these contextual factors.

Different comparative risk perception studies have been conducted in order to observe the influence of contextual factors on the VSL, see e.g. Chilton et al., (2002); Chilton et al., (2006); Jones-Lee and Loomes (1995), Tsuge et al., (2005) Jones-Lee et al., (1985) . The evidence has been mixed as to whether individuals' perceptions of these risk characteristics significantly affect the valuation of mortality risk.

Does age matter?

The issue of age has been analysed within the life-cycle consumption models taking into account how the valuation of a change in mortality risk varies with the life-cycle consumption. The paper demonstrated that when taking life-cycle consumption into account we cannot predict how VSL_j changes with age. This results is in accordance with the theoretical analyses made in (Johansson 2002), which predicts that the VSL may increase, be constant, or decline with age or follow an inverse U-form depending on assumption regarding utility discount rates, hazards rates, and optimal rates of consumption. An inverse U-form describing the relationship between WTP and age has been found in life-cycle consumption models (Shepard and Zeckhauser, 1984) based on analyses of life-time earnings in the U.S., an assumption of no available markets for trading (the so-called Robinson Crusoe case) and a time-invariant utility function assumption. If on the other hand the empirical analyses in the life-cycle models are carried out assuming a perfect market a decreasing relationship is found.

Methods

This paper reports results from a web based stated preference survey on VSL in two different contexts; traffic and health. The survey was carried out in Denmark in July 2007 using an internet panel. Respondents were initially asked some introductory warm-up questions related to their own health, health behaviour as well as about their awareness of specific types of risks (health and traffic) and general propensity to take on risks or finding safety very important in all aspects of life. Each respondent was subsequently randomised to two different versions of the questionnaire (traffic and health). All respondents were asked to value an intervention of a private nature which could reduce the probability of death. Respondents in both arms were presented with a visual diagram illustrating the risk reduction. The diagram was similar in format to the diagram used in Krupnick et al. (2002).

In arm A the respondents were told that the risk reduction could be obtained by buying a new security system for their car. The intervention in the traffic arm offered a reduction in mortality risk of traffic accident from 2 out of 1000 to 1 out of 1000 over a period of 10 years. In the health arm the respondents were offered a reduction in mortality risk of cardiovascular diseases from 2 out of 1000 to 1 out of 1000 over a period of 10 years. The respondents were told that the risk reduction could be obtained by way of medication. This involved taking one pill a day, which involved no side-effects or increased risks of other diseases. In both settings the risk reduction was presented as the individual's personal risk.[2] The magnitude of the 10 year base-line risk in both arms corresponds to the true average risk faced by a 40 year old in Denmark.

The contingent valuation exercise involved an initial dichotomous choice exercise where respondents were randomised to one of the following annual price bids (which were to be paid each year over the next 10 years): 120, 600, 1.200, 3.000, 6.000, 9.000 or 15.000 DKK (1 DKK = 7.5 €). Subsequently, all respondent were asked to state their maximum WTP with the use of a payment card on which a list of values ranging from zero to more than 15.000 DKK were presented.

Results

A sample of 1041 Danish individuals in the age-group 18 to 80 years old were interviewed online. WTP estimates from a two-sample split are compared and VOLY's are derived. Based on the results in the present data set, we find that the VSL is context dependent. Overall, we find a large variability in the VOLY estimates with respect to age. In the context of health we find a mean VSL of DKK 40 million; increasing with age. The corresponding age-specific VOLYs are found to be increasing from DKK 500,000 to DKK 4 million (mean of DKK 1.3 million). In the traffic context, the mean VSL is found to be DKK 24 million. With regard to age, an inverse U-form is found in the OLS regressions yet in the OLS_{log} and in the logit a linear increasing relationship is found. The age-specific VOLYs in the traffic context are in the range from DKK 300,000 to DKK 1 million (mean of DKK 800,000). Compared to previous VSL results based on stated preference studies, the VSL and corresponding VOLYs found in this study are relatively high (see Braathen et al., (2009)). Our results, however, are in keeping with a previous stated preference VSL study of the Danish population estimating mean VSL to be in the range of DKK 33-69 million (Kidholm 1992) which is well above the values derived in the current study.

Conclusion

This paper presents results from a web-based stated preference survey on VSL. WTP estimates were elicited for reductions in the risk of dying for two causes; 1) traffic accident, and 2) cardiovascular diseases. We find variability in the VSL estimates according to contexts. This raises concern about using a traffic-based VSL to derive a VOLY which could be used in the economic evaluation of a health care program. Moreover, given the existing formulas for calculating a VOLY, we find a large variability in the VOLYs across age-groups and hence we reject the notion of proportionality between VSL and life expectancy. In the health context very high VOLYs have been

[2] The probability was held constant across settings in order to be able to compare the influence of context on VSL in isolation

found for the individuals above the age of 70. Since many health care services are directed towards this specific age group it is not a trivial question to establish whether the relatively high VOLY found for this age group is in fact an accurate reflection of their preferences. Alternative explanations could be decreasing marginal utility, income as an insufficient proxy for wealth, and problems related to VSL as elicitation methodology. Scope for future research is to disentangle these possible reasons for observed variation in preferences in order to verify whether the preferences are valid for policy making.

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The role of discounting in socio-economic analysis and CO2 damage cost estimation

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This paper analyses how the theory on discounting and valuation of CO2 damage costs can be made operational and be implemented in the socioeconomic assessment of energy projects.

Three central problems in discounting are identified:

- Problem 1: Intergenerational and geographic inequality
- Problem 2: Non-substitutability of natural capital
- Problem 3: Uncertainty

Using a literature survey (Stern (2007), Azar (1999), Johansson-Stenman (2005), Sterner og Persson (2008), Cropper and Laibson (1999), Fisher and Narain, (2003), Dietz (2006) and others), the impact of these issues on the CO2 damage costs are quantified. We find that CO2 damage costs can be 5-8 times higher than the current market price for CO2-quotas; however, the damage is possibly still underestimated, since no study takes account of all three problems, but only takes account of one problem at a time.

Furthermore, the discount rate used for discounting when making cost benefit analyses of entire energy projects (the “outside” discount rate) is less significant than the discount rate used when defining the CO2 damage costs entering the cost-benefit analysis (the “inside” discount rate).

Research is still needed to combine the three central problems in one CO2 damage cost estimate.

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Relation between burden of disease and cost of illness

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Introduction

Reliable health data and statistics are the foundation of health policies and strategies (WHO, 2008). The WHO has collected a vast dataset on the physical burden of disease (BOD). But, the BOD concept alone does not give an indication of the economically justifiability of measures in the health system (Hofstetter and Müller-Wenk, 2005). This paper clarifies the relation between the BOD and the related economic cost of illness (COI). This may contribute to a more optimal allocation of resources in the struggle of reducing the BOD. More concrete, it can help to answer questions like: "What are the most efficient strategies to reduce the BOD and COI simultaneously?" and "In what range are investments to reduce the BOD beneficial for society?". For policy makers this should be very relevant in prioritizing public health spending. More insight in the COI-BOD relationship could be useful to transfer data for a specific type of disease from one indicator to the other (Melse and De Hollander, 2001) and more consistency could be realized in the communication between different groups of stakeholders.

Methodology

Burden of Disease (BOD) expressed in Disability Adjusted Life Years (DALYs)

The BOD expressed in DALYs for a disease or injury are calculated as the sum of the years of life lost (YLL) due to premature mortality in the population and the years lost due to disability (YLD): DALY = YLL + YLD. YLL are calculated by multiplying the number of deaths at each age with a global standard life expectancy at that age. YLD for a particular cause in a particular time period are basically estimated as follows: YLD = I x D x L, where I is the number of incident cases in the reference period, DW is the disability weight on a scale from 0 (perfect health) to 1 (death), and L is the average duration of disability measured in years.

The term disability in this context refers to loss of health, conceptualised in terms of functioning capacity in a set of domains such as mobility, cognition, hearing and vision (WHO, 2008). The original GBD study established disability weights for approximately 500 disabling sequelae[3] of diseases and injuries, in a formal study involving health workers from all regions of the world. Distributions could vary by age group and gender across disability classes and were estimated separately for treated and untreated cases where relevant (Mathers et al., 2003, 2006; WHO, 2008).

For this paper, data on the BOD (DALYs=YLL+YLD) were selected from the WHO for the year 2004 for the relevant diseases for Germany (WHO, 2008) (cf. Table 1).

[3] A sequela is a pathological condition resulting from a disease, injury, or other trauma.

			COI - data			WHO - data								
ICD-10 code	GBD -code	Disease	Total costs (mio €)	DMC (mio €)	Prod. Loss (mio €)	DALY (years)	YLD (years)	YLL (years)	Deaths	I Incidence	P prevalence	DW (x 0,01)	average L (days)	COI/BOD (€/DALY)
H90, H91	W102	Hearing loss, adult onset	1124	896	228	396582	396582	0	0	286377	20087871	8	6240	2834
G30	W087	Alzheimer and other dementias	1242	1185	57	401914	373698	28217	8904	231237	1050155	64	924	3090
C25	W066	Pancreas cancer	1278	309	969	94588	2451	92137	13424	13417	28970	24	282	13511
H40-42	W099	Glaucoma	486	429	57	34808	34808	0	0	15297	254122	60	1384	13962
J40->J44;+J47	W112	COPD	5486	2693	2793	353664	241379	112285	22931	91054	1519301	39	2494	15512
I20-I25	W107	Ischaemic heart disease	13714	6133	7582	882336	125699	756637	169420	240440	1228712	22	866	15543
C33-34	W067	Trachea, bronchus, lung cancers	5291	1073	4218	336042	9593	326449	40922	43628	159840	15	549	15745
F32-34	W082	Unipolar depressive disorders	12404	4139	8266	765653	765263	390	46	2795071	2239640	30	331	16201
K51	W116	Peptic ulcer disease	435	207	228	26684	9078	17606	3701	99021	357858	0	10141	16302
C18+C20	W064	Colon and rectum cancers	3774	1665	2109	228545	39005	189540	31423	79752	455194	22	823	16513
C16	W063	Stomach cancer	1602	405	1197	89922	3687	86235	12968	14935	54843	22	415	17815
C53	W070	Cervix uteri cancer	588	132	456	32804	6953	25851	2692	11190	72729	7	3030	17925
J45-J46	W113	Asthma	2256	1572	684	112841	97178	15663	2210	170295	3008312	6	3547	19993
C67	W074	Bladder cancer	850	508	342	38906	7900	31006	6243	20780	284226	9	1614	21848
C50	W069	Breast cancer	4617	1596	3021	199113	32978	166135	20075	70280	958280	9	1996	23188
C61	W073	Prostate cancer	1839	1269	570	74495	16516	57979	12279	38346	158287	13	1171	24686
E10-14	W079	Diabetes	7095	5100	1995	280671	167125	113546	24541	1133771	7059333	3	1645	25279
C91-95	W076	Leukaemia	1779	639	1140	61413	2742	58672	7975	9962	67664	9	1110	28968
C81-90; C96	W075	Lymphomas, multiple myeloma	2326	901	1425	77276	4154	73121	10763	14623	135523	6	1828	30100
I60-I69	W108	Cerebrovascular disease	12432	7929	4503	407828	116491	291337	73464	191244	725962	23	963	30483
C43-44	W068	Melanoma and other skin cancers	953	383	570	28761	1904	26857	3168	9532	165616	5	1618	33136
C00-C14	W061	Mouth and oropharynx cancers	1992	339	1653	56050	3585	52465	5039	7594	46885	9	1921	35539
A15-A19	W003	Tuberculosis	222	108	114	5229	1822	3408	455	6078	5026	28	391	42453
G40-41	W085	Epilepsy	2392	1138	1254	55583	30857	24726	1965	55709	413000	6	3346	43035
B20-B24	W009	HIV/AIDS	749	122	627	15962	5348	10614	547	2377	48104	32	2581	46924
F20	W084	Schizophrenia	6998	1811	5187	106866	105842	1023	91	8212	404137	35	13410	65484
B16	W018	Hepatitis B (g)	421	22	0	3716	479	3237	360	9023	7692	20	99	113292
I10-15	W106	Hypertensive heart disease	9678	8025	1653	82981	15583	67398	19444	54738	108289	20	520	116630
K02	W144	Dental caries	7577	7577	0	38100	38100	0	0	21958861	439177	8	8	198874

Table 1: The 29 endpoints ordered by increasing COI/BOD. Data for Germany in 2004; sources: WHO (2009); GISFHM (2009) and calculations VITO.

Cost of Illness (COI)

COI are calculated as the aggregation of direct and indirect costs (EPA, 2006). The direct costs are immediately related to the treatment of the illness and can be subdivided in medical and non-medical costs. The costs that are indirectly related to the disease can theoretically be subdivided in loss of productivity, leisure time loss and costs related to suffering (physical and/or psychological) also referred to as intangible costs. Our source of COI data was the German information system of Federal Health monitoring, further referred to as GISFHM (2009), reporting direct medical costs and Lost Workforce Years (LWY) for 29 diseases in Germany. Direct medical costs per disease are based on a top down division and attribution of the total national costs for medical infrastructure and activities to the specific diseases. LWY are a good indicator for productivity loss. The calculation of LWY was related to 3 possible causes: (1) (temporarily) inability to work calculated on a prevalence basis for 2004, (2) (permanent) invalidity and (3) premature death, both calculated on an incidence basis for 2004. These were calculated for the potential workforce population, aged between 15-64 years. By multiplying the LWY with 57004 euro, the average monetary output per worker in Germany in 2004 (OECD, 2005), the productivity loss was calculated and this for different illnesses. Total COI (cf. Table 1) was calculated as the sum of direct costs and monetised productivity loss.

Literature review

German COI-data from 14 selected literature studies were also examined versus corresponding BOD data from the WHO (data year 2004) for different diseases in Germany (Figure 2). Findings were similar as with the GISFHM (2009) data. Because of lack of space these data are not discussed here. For more details we refer to Aertsens et al. (2010) and Buekers et al. (in prep.).

Findings

Correlation between COI and BOD

Data on COI (total costs) for different disease classes and subclasses from the GISFHM (data year 2004) were compared with corresponding BOD data reported by the WHO (data year 2004). The positive correlation between BOD and COI (Pearson corr. coeff.= 0.75) (cf. Figure 1) was significant at the $P<0.001$ level

A wide range in COI/BOD depending on the disease

Results for COI/BOD data per disease are given in Table 1. In this analysis the COI/BOD ranged from 2834 euro/DALY (hearing loss) to 198874 euro/DALY (skin diseases in general). Thus a difference of a factor 70 in COI/BOD across all considered diseases for Germany. This finding means that even within one country it is not possible to have a uniform translation of DALYs into euro by applying one conversion factor for all diseases. The translation of burden of disease into cost of illness will differ depending on the disease in question.

In order to get a better insight in the COI-BOD-relation, Direct Medical Costs, Productivity loss and total COI, were studied separately in relation to incidence, number of deaths, YLD, YLL, DW, L, ... (Aertsens et al., 2010). The findings are summarised in the following three sections.

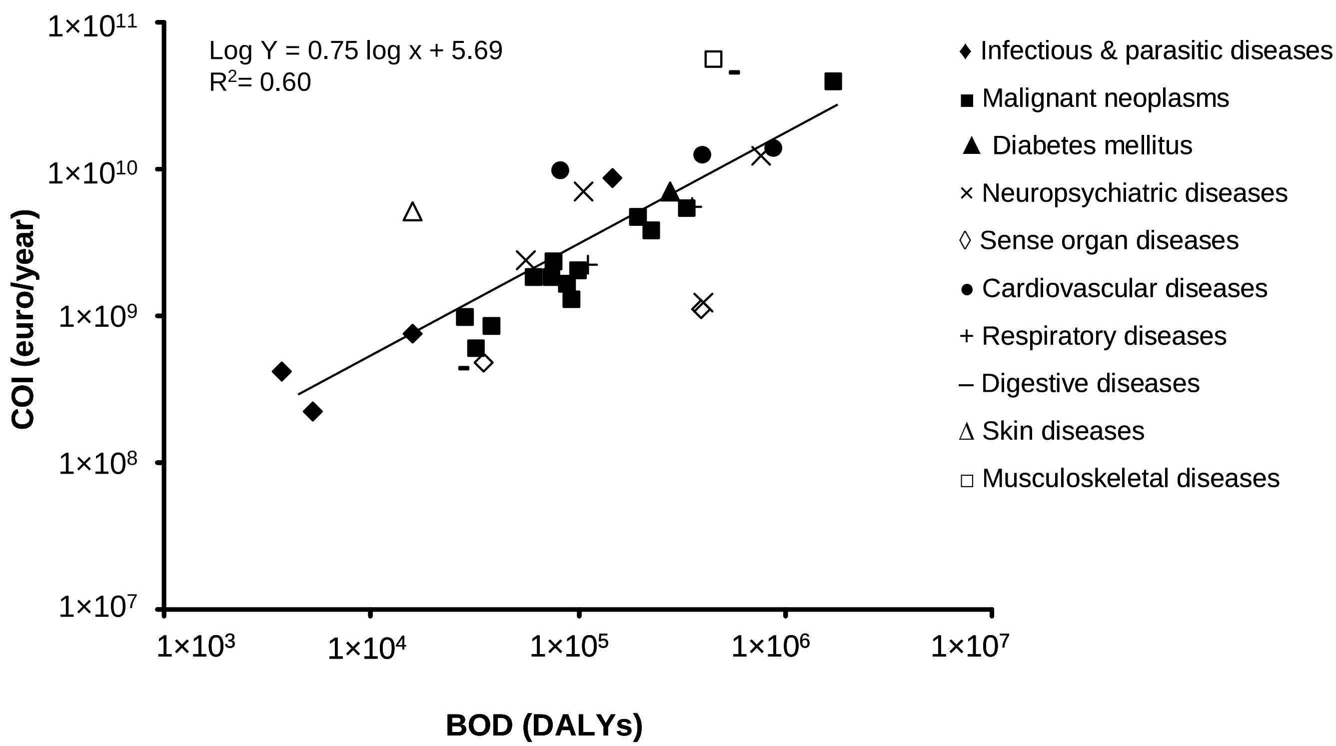

Figure 1: Cost of Illness (GISFHM; 2009) versus Burden of Disease (WHO, 2008) for Germany in 2004.

Direct Medical Costs and the functional relation with WHO data

It was found that the Direct Medical Costs (DMC) are positively and significantly correlated with Years Lived with Disability (YLD), with the number of deaths, with the incidence (I) and prevalence (P) of the disease and the duration of the disease (L). Contrary to the expectation no positive correlation was found between DMC and the Disability Weight of (sequelae of) diseases. For more details cf Aertsens et al. (2010).

A multiple linear regression model was composed to estimate the DMC based on data provided by the WHO (2009). The model has a R^2 of 0,60 (R^2-adj.=0,56); β_1, β_2 and β_3 were all significantly different from 0 (α<0,05). β_0 was not significantly different from 0, which makes sense.

$$DMC_{(i)} = \underbrace{-2.915.676.000}_{\beta_0} \underbrace{+472.000}_{\beta_1} \text{ x } \text{“Incidence}_{(i)}\text{”} + \underbrace{467.000}_{\beta_2} \text{ x } \text{“deaths}_{(i)}\text{”} + \underbrace{311.000}_{\beta_3}\text{xLOG10}(YLD_{(i)})$$

Productitivity Loss as a function of YLL and YLD

LWY and the related Productivity Loss (PL) caused by a disease can also be estimated based on the WHO-data. The following model proved to be highly significant (at 0,001 level) and has a R^2 of 0,84 (R^2-adj.=0,83). β_1 and β_2 were all significantly different from 0 (α<0,05). While β_0 was not significantly different from 0, which makes sense. As the PL based on a HCA for Germany in 2004 can be approximated by multiplying the LWY-total with 57004 euro. A similar model can be obtained for Productivity Loss.

$$LWY\_total_{(i)} = \underbrace{4.137}_{\beta_0} + \underbrace{0,707}_{\beta_1} \text{ x } \text{“}YLD\_emp\_up\_to\_60_{(i)}\text{”} + \underbrace{0,666}_{\beta_2} \text{ x } YLL \text{ “}YLL\_emp\_up\_to\_60_{(i)}\text{”}$$

Estimating the total COI as a function of the WHO-data

To verify the relation between total COI and the WHO country data, the following model was tested on our German 2004 dataset. It was significant at 0,001 level, has a R^2-adj. of 0,73 (R^2 of 0,76) thereby on average explaining 73% of the total COI of a certain disease, based on the independent BOD components of a the disease provided by the WHO (2009). All βs were significant at 0,01 level.

$$TOTAL\ COI_{(i)}\ (mio\ €) = \underbrace{1400}_{\beta_0} + \underbrace{0,273}_{\beta_1} \text{ x } \text{“}Incidence_{(i)}\text{”} + \underbrace{0,682}_{\beta_2} \text{ x } \text{“}deaths_{(i)}\text{”} + \underbrace{0,501}_{\beta_3} \text{ x } \text{“}YLD\_emp\_up\_to\_60_{(i)}\text{”}$$

Conclusion and discussion

Based on data for 2004 from 29 diseases in Germany a significant (P< 0.05) correlation between COI and BOD was found (Pearson correlation coefficient 0.75). However, the COI/BOD expressed in euro/DALY varies with a factor 70 across all considered diseases for Germany in 2004. This means that even within one country it is not possible to have a uniform translation of DALYs into euro by applying one conversion factor for all diseases. The important differences in the COI/BOD-ratio between diseases may have important implications for policy makers. When optimizing health expenditures, policy makers are advised to minimize the BOD, under a certain budget constraint or in other words to maximize the QALY/euro ratio. Our findings imply that the reduction of 1 DALY for certain diseases will result in a much lower reduction in the “societal cost” than for other diseases. This raises the question: “Should policy makers also take into account the reduction of COI, when optimizing the health expenditures. And how to trade off between minimizing the BOD and minimizing the societal cost of diseases? From a (macro) economic point of view, there is an argument to prioritize the minimization of the COI as this will lead to an increased future welfare, which will in turn positively influence the future government budget. The trade off also has ethical implications. Giving priority to minimizing the COI rather than the BOD, will prioritize expenditures beneficial for the young and active population, while providing less resources for illnesses that rather affect “non-productive” people. In the long term however, theoretically it might also be beneficial for the “non productive” population as higher “welfare” may lead to higher “well being”.

To improve insight in the COI-BOD relation, the relations between the individual components were studied. It was found that the Direct Medical Costs (DMC) are positively correlated with Years Lived with Disability (YLD), the number of deaths, the incidence (I) and prevalence (P) of the disease and the duration of the disease. Contrary to the expectation no positive correlation was found between DMC and the Disability Weight. It was also found that Lost Workforce Years and the related Productivity Loss caused by a disease can be estimated significantly based on the WHO-data.

Combining these findings it was also possible to estimate the order of magnitude of the societal cost of illnesses based on data provided by the WHO. Seen that there is a vast set of data on the BOD, this is an important source of information that is currently insufficiently used.

Further research could certainly improve the insights in the relation between BOD and COI. For more details we refer to Aertsens et al. (2010).

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Future costs in cost-effectiveness analysis: an empirical assessment

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Aim: The aim of this study was to assess the impact on the cost-effectiveness ratio of including measures of production and consumption following a health care or health promotion intervention that improves survival.

Data and methods: We defined the net incremental consumption, or future costs, as the change in consumption minus change in production, while differentiating between health care and non-health care consumption. Based on 2005 register-based data for the entire Danish population, we estimated the average value of annual production and consumption for one-year age groups. We computed the net consumption in the remaining expected lifetime and the net consumption per life year gained for different age groups.

Results: Age has a profound effect on the magnitude of net consumption. When including net incremental consumption in the cost-effectiveness ratio of a health care or health promotion intervention, the relative cost effectiveness changed up to €21,000 across age-groups. The largest difference in the cost-effectiveness ratio was observed amongst the 30-year olds where costs were reduced significantly due to significant future net contributions to society.

Conclusion: This paper contains cost figures for use in cost-effectiveness analyses, when the societal perspective is adopted and future consumption and production effects are taken into account. The net consumption varies considerably with age. Inclusion of net incremental consumption in the cost-effectiveness analysis will markedly affect the relative cost-effectiveness of interventions targeted at different age groups. Omitting future cost from cost-effectiveness analysis may bias the ranking of health care interventions and favour interventions aimed at older age groups. We used Danish data for this assessment, and our results will therefore not represent true figures for other countries. We do, however, believe that the overall impact of including net production value in CEA will be similar in other countries that have similar transfers of income from the younger age-groups to older age-groups as well as publicly financed social and health care services.

Keywords: future costs, cost-effectiveness analysis, life years saved

Health Services Costs of Treating Patients with Air Pollution Related Illnesses

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Background

Individual use of health care services is an important component in the description of resource use and cost associated with air pollution. Changes in the pollution level might influence both the incidence of new episodes of illness (e.g. lung cancer), the prevalence of illness (e.g. asthma), and the cost of treating the illness (e.g. through severity and complications). Changes in the prevalence might cause changes in the need and demand for health care services. Demand changes will – ceteris paribus - change the production of health care and thus the total cost of the health care services. In a simple analytic framework such cost consequences for the whole health sector could be modelled as the change in demand for services (e.g. number of patients who require lung cancer treatment) multiplied with the long-term average cost of a newly diagnosed lung cancer patient.

Objectives

In this analysis we will develop and test a method to establish the *unit costs* to be included in a cost function for health care sector cost. Initially, we base the method on an assumption that the incremental costs of treating an additional patient equate the long-term average cost.

Method

In this analysis we include a limited number of illnesses (lung cancer and Acute myocardial infarction (AMI)) as illustrative examples. We identify individuals who have received hospital care for these diseases through inspection in a 30% sample from the National Patient Registry (2004-2008). We identify relevant diseases through the International Classification of Diseases (ICD-10) (Lung cancer: C34; AMI: I21). In order to identify new cases we apply a 'wash-out' period of 2 years (2004-2006) during which the individuals should not have had any hospital contact with the diagnose. We thus define individuals who have a hospital contact with a relevant diagnostic code as patient with an episode of the disease.

For each individual in these groups we analyse their consumption of health care services (primary care and hospital care) using all records of health care services provided. We count the number of services provided by different specialists in the primary care and the number of inpatient admissions, contacts in accident and emergency department and clinical outpatient departments. To cost the services we apply the fees paid to the providers by the social health insurance and the national Diagnostic Related Groups for hospital services. We apportion the resource consumption and cost in three months periods starting from the first hospital episode. We will thus be able to describe the cost in the first 24-36 months after the first hospital contact.

To describe the cost attributable to illnesses associated with air pollutant we estimate similar costs for a group of individuals who have not had the disease. By stratified analysis for different age and gender groups we will be able to derive a cost estimate that represent the average health care cost associated with the diseases for specific gender, age groups and for different stages (time periods after diagnosis) of the disease.

Session 4: Integrated Modelling

Keynote speaker: Fabian Wagner, IIASA, Austria

Chair: Steen Solvang Jensen

Regional Air Quality Assessment of Biofuel Scenarios in the Road Transport Sector

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Biofuels, especially ethanol, have been in political focus for the last couple of years, both internationally and in Denmark. For the European Union, the EU Biofuel Directive sets indicative targets for the biofuel share of transport energy demand in each Member State of 2% by the end of 2005 and 5.75% by 2010 (European Union, 2003). Three liquid biofuel types have mostly been used as transport fuel: biodiesel, ethanol and methanol (Edwards et al 2006, IEA 2004). Here, biodiesel is used as a generic term for biofuels suitable for diesel engines, while the remaining two fuels are used in gasoline engines. There is strong growth in the use of biofuels globally, particularly in Brazil and the USA, as well as in demand for technological development and marketing of both vehicles and fuels, especially ethanol. In Germany biodiesel is widely used, mainly in order to secure supply of fuel for transport.

In this study a scenario approach is use to assess the impact biofuels have on the environment and human health in Denmark. Three different biofuel road transport emission scenarios at different years from 2004 to 2030 and barrel prices (65$ and 100$) has been applied:

1. A business-as-usual scenario, use of 0% biofuel in the road transport sector.
2. The biofuel share will grow to beyond 8% by 2030, in line with the Commission's estimate of biomass potential for transport fuels for the Biofuel Directive (CEC, 2001).
3. The biofuel share of the road transport energy will grow to reach 25% in 2030, i.e. the EC target of 20% alternative fuels is met entirely by biofuels (CEC 2001).

That means the total of 24 different emission scenarios (Winther, 2009) is applied in this study. To model the abovementioned scenarios the integrated model system, EVA (Economic Valuation of Air pollution) is used. EVA is based on the impact pathway chain and consists of the Danish Eulerian Hemispheric Model (DEHM, Christensen, 1997; Brandt et al., 2001; Frohn et al., 2002), address-level or gridded population data, state-of-the-art exposure-response functions and monetary valuation of the impacts from air pollution.

In the work presented here a brief introduction to the EVA system will be given, followed by a presentation of the influence the road transport sector will have on the environment if biofuls are not added to transport fuels from 2004 to 2030 (business-as-usual) and if the biofuel share of transport energy is 25%. Finally, an assessment of health-related economic externalities of the air pollution from these two cases will be given.

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Integrated assessment of the introduction of biofuels in the Danish Transport sector

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Introduction

During the last decade procedures and processes to support the integration of environmental issues in sector policies and sustainability issues in EU policy making have been developed to ensure increased policy coherence. Better regulation at EU and member state level is requested, and ex-ante impact assessment of policy alternatives is one of the methods implemented in the Commission policy making procedures, and is also implemented in some, but not all, member states (CEC, 2005). National ex-ante impact assessment systems have been implemented, but do not always give proper consideration of environmental aspects (Raggamby *et al*., 2007).

Targets for the introduction of biofuels in the transport sector in Denmark is now a 5.75 % share of fuels in 2010, phased in until 2012, and in 2020, 10% of the energy consumption in the transport sector should be covered by renewable energy. This paper presents the approach developed and some results from an integrated assessment of the introduction of biofuels in the transport sector in Denmark to reach these targets using biofuels. Based on forecasts for road traffic increase until 2030, estimations of road traffic energy demand, biofuel and biomass demand under different policy targets for biofuel mix are estimated. Options for meeting this demand under a strategy of self-sufficiency are discussed.

Background

Energy security and climate change considerations, and the subsequent need for decoupling from fossil fuel dependency, have targeted the production of energy from biomass as an important source for substitution. EU has through the Renewable Energy Directive (CEC, 2009) laid down targets for the use of renewable energy to reach 20% of the consumption in 2020, while the transport sector must reach a share of 10%. Recent legislation already supports this conversion in many countries, and incentives for national production exist. Required shares of biofuel mix lie in the range of 5-20 % to be reached at different years until 2020 (Petersen, 2008, pp. 389-390, Cushion et al., 2010, Ravindranath et al., 2009).

Biomass conversion to biofuels for transport is however heavily debated. The relatively low energy efficiency due to loss in the transformation process, in comparison to the use of biomass as fuel in combined power and heating plants has been highlighted (Teknologirådet, 2009), but it has been argued that side products from biofuel production, e.g. animal feed, and other side products should also be included in the calculations of energy efficiency. Another issue is the land competition which may result from land requirements exceeding the availability of land, which may threaten food security and impose indirect land use change (iLUC) with possible unwanted environmental impacts, such as loss of tropical forests (Cushion et al, 2010), and the ensuing carbon dept (Fargione, 2008, Searchinger 2008 and 2009). Yet another consideration is the environmental impact of biomass production from direct land use changes and crop substitution, which may be both positive and negative. Integrated studies, such as LCA studies, however, often focus on the greenhouse gas balances of biofuel conversion (Menichetti & Otto, 2009), while other environmental impacts are to some extent overlooked (Bringezu *et al*., 2009). Approaches and methods for ex-ante integrated assessment of policy proposals to implement national bioenergy targets and to include land use change and environmental impacts are needed (Petersen 2008).

Fischer et al (2010) produced land use scenarios for Europe including Ukraine, anticipating up-keeping of current levels of self-sufficiency in food and feed production. They found the total availability of land for biofuel feedstock in 2030 to be between 44 million. ha and 72 million ha, depending on the environmental considerations and the priority to energy production (in 2000 the total cultivated land and pasture in Europe plus Ukraine was about 240 million ha). In this land use scenario Denmark figured with 289.000 ha for biofuel feedstock in 2030. The Danish agricultural organisation estimates a target of 100.000 ha cultivated with perennial energy crops in 2020 (and an additional 1 mio. tons of straw) (Landbrug og Fødevarer, 2009). A Danish study on behalf of the Danish Ministry of Food, Agriculture and Fishery (2008) estimated that the app. 100.000 ha used mainly for rape production in 2007 could be increased to 125.000 ha, and that 224.000 ha of other cropland could also be used for energy purposes. In addition, 57.000 ha set-aside and 115.000 ha low-lying grassland was included in the biomass for energy estimation, as well as 626.000 ha straw from crops and rape.

Data and methods

The overall method used is to compare scenarios for biofuel use in the road transport sector and the related demand for fuel production and agricultural products to a reference, where no biofuel is used. Consequences are quantified as differences between scenario and reference. Emphasis is on building the assessment methodology and system delimitations, while only using mainstream crops in the assessments. Two scenarios are produced – one compliant to the policy targets (HS1) and another reaching a 25% share in 2030 (the high scenario, HS2).

Reference forecasts

Two types of forecasts are used as reference for the impact assessments. The first is a forecast of the road traffic until 2030, while the second is a forecast of the agricultural area and the related land use in the same period.

Transport forecast. The forecast used in this study (Jensen & Winther, 2009) is based on a forecast made for a Danish Infrastructure Commission during 2006-2007 (Infrastrukturkommissionen, 2008). Two forecasts were set up based on alternative assumptions on oil prices, a low price assumption of 65$ pr barrel and one of 100$ pr barrel (the base case). The model used in the forecasts is a Danish econometric model. It uses the relationship between income (GDP) and variable costs on the one hand and car stock and annual mileage of a car on the other, and it is based on the historical relationship between these. The price of fuel is about 60 % of the variable costs. The projected traffic is estimated by multiplying the stock by the annual mileage per car. Forecasts for busses and lorries is produced, based on constant yearly growth in traffic. Yearly growth is related to GDP and is based on analyses of the relationship between economic growth and traffic. Growth in GDP was set to 1.2 % pr year on average following the Danish Ministry of Finance (2005). In the forecast with an oil price of 65$ pr barrel, traffic is estimated to grow by 1.4% p.a. 2005-2030 for cars and vans, 2.2% for lorries and 0.0% for busses – on average 1.4%. In the base case with an oil price of 100$ pr barrel traffic growth is reduced to 0.8% a year. From these traffic estimates, energy consumption is estimated. It was assumed that no increase in fuel efficiency would take place[4]. Fuel consumption is thus directly related to traffic work. The relative share of diesel and petrol cars was incorporated, as well as the effect of the fuel efficiency in already sold cars. The resulting energy demand is shown in table 1 for the 100$ forecast, which is the forecast that will be used for the calculations of land use demand.

100 $ forecast	Fuel consumption TJ				
	2010	2015	2020	2025	2030
diesel	105.801	120.326	131.986	142.441	152.300
gasoline	55.211	48.599	47.837	48.695	50.213

Table 1: Estimated fuel consumption, based on road traffic forecast, assuming oil price, 100$

Agricultural land use reference scenario. This reference scenario (Larsen *et al*, in prep) is based on following presumptions: the agricultural land area will be reduced according to known reductions due to political environmental agreements (e.g. on afforestation and set-aside) and on estimated reductions due to development of urban areas and infrastructure. Moreover, the livestock/dairy sector develops as foreseen by the sector represented by the Agricultural organisation until 2015, and it stabilises after this towards 2030. This results in a reduction of the cultivated areas from 2.644.700 ha in 2008 to an estimated area in 2030 of 2.465.600 ha, i.e. a reduction of approximately 200.000 ha. The estimated area for roughage is almost constant in the period, while areas for concentrates increase. Consequently,

[4] A number of arguments support this assumption, including that fuel efficiency is implicitly incorporated in the price elasticity for small cars.

the area for feed production takes up an increasing share of the cultivated area, and with a decreasing total agricultural area the remaining area – now used for fallow, high value seeds, regional products, and other crops are decreasing from 577.000 ha in 2008 to 267.000 ha in 2030. Given that high value seeds and regional products are continuously competitive, and that the present fallow area represents areas that will no be cultivated, the resulting "free" area now used for other crops descends from 371.000 ha to 107.000 ha in the scenario period.

Scenarios for introducing biofuels in the road transport sector

The policy scenario (HS1) implements the targets of 5.75 % biofuel use in 2010 and 10 % in 2020. The high scenario (HS2) use biofuel shares of 15 % in 2020 and 25 % in 2030. Second generation bioethanol are phased in from 2010, as illustrated in figure 1.

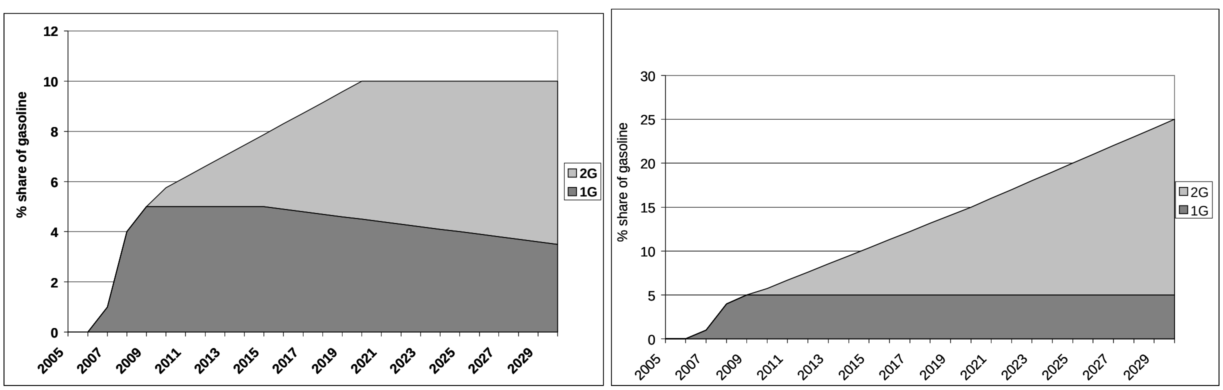

Figure 1: HS1 and HS2, bioethanol demand distributed on 1. and 2. generation conversion technology

The assumptions behind the development of the scenarios are:

- The shares of bioethanol and biodiesel are the same (implying that, as the number of diesel cars increases more than petrol cars, the biodiesel demand will increase more than the bioethanol demand.
- The agricultural products used are wheat (kernels) for bioethanol and rape (RME) for biodiesel, while wheat straw is used as raw material for second generation bioethanol production.

Figur 2a and 2b shows the relative distribution of energy from biodiesel and bioethanol respectively, for the two scenarios.

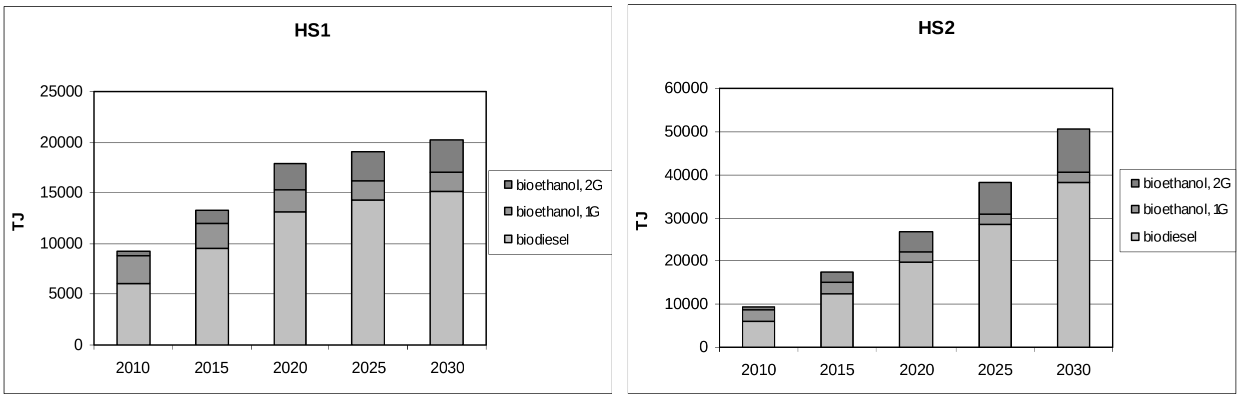

Figure 2: Distribution of biofuel shares in biodiesel, 1G bioethanol and 2G bioethanol

Results and discussion

Based on the estimated biofuel demand, the biomass demand and resulting land claims are estimated. In these estimates, it is anticipated that only 50% of the straw is used pr area unit, allowing for present uses of straw. The results are shown in table 2 and table 3.

table 2	HS1 area demand 1000 ha				
rape	129	202	280	302	323
wheat	44	38	34	31	28
straw	18	60	112	124	139

Table 2: land demand in HS1

table 3	HS2 area demand 1000 ha				
rape	129	265	420	604	808
wheat	44	39	38	39	40
straw	18	112	204	310	426

Table 3: Land demand in HS2

The wheat and straw demand in HS1 could be met within the present land use, given that the "free" area is used for biofuels such that 25% (as much as rotation allows) is taken up by rape, and the remaining area is used for wheat and straw. Then only 8000 ha in HS1 and 20.000 ha in HS2 need to be found in the present wheat area, which takes up around 700.000 ha. Straw demand can also be met within the total wheat area. If the present rape area is used solely for biofuel, and 25% of the "free" area is used, land for rape will still be insufficient before 2020.

If the development in the car park allowed that this deficit could be covered by bioethanol, and given the same distribution among 1st and 2nd generation bioethanol, this would demand additional 20.000 ha wheat in HS1, and a total of 247.000 ha straw, which is fully available.

For HS2 the area for rape is almost three times as large, and if the deficit should still be covered by bioethanol this would claim around 30% of the present wheat area, while the straw would take up 80% of all wheat straw or 80% of the straw from all cereal production, given the 50% rule is still applied.

These estimates have not yet included the fodder, which can be retrieved as a side product from the bioethanol production and which will substitute some of the fodder demand, and thereby release more land for other production. Moreover, technological improvements and productivity increases can be expected. Eventually other agricultural strategies with less dominance of dairy and meat production would also free areas for e.g. biofuels.

Conclusion

The scenarios presented show that the policy targets for biofuel implementation cannot be fully met for the RME part. If 25% the diesel share could be interchanged to bioethanol, it would be possible to realise in terms of land availability, given that 2G technology is soon available, and under present agricultural strategies. Higher share of biofuel use will need to be based on higher shares of bioethanol to RME fuels or on larger changes in agricultural strategies.

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Integrated Well-to-wheel assessment on biofuels, analysing Energy, Emission and Welfare economic consequences

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Introduction

Various biofuel evaluation methods exist, with different analytical framework setup and different scopes. The scope of this study is to develop an integrated method to evaluate the consequences of producing biofuels. The consequences should include energy consumption, emission and welfare economic changes within the well-to-wheel (WTW) flow chain focusing on the production of biomass, and the subsequent conversion into bio fuel and combustion in vehicles. This method (Møller & Slentø, 2010) is applied to a Danish case, implementing policy targets for biofuel use in the transport sector and also developing an alternative scenario of higher biofuel shares.

This paper presents the results of three interlinked parallel running analyses, of energy consumption, emissions and welfare economics (Slentø, Møller & Winther, 2010), and discusses the feasibility of those analyses, which are based on the same consequential analysis method, comparing a scenario situation to a reference situation. As will be shown, the results are not univocal; example given, what is an energy gain is not necessarily a welfare economic gain.

The study is conducted as part of the Danish REBECa project. Within this, two main scenarios, HS1 and HS2, for biofuel mixture in fossil diesel fuel and gasoline are established. The biofuel rape diesel (RME) stems from rape seeds and bioethanol stems from either wheat grains (1st generation) or straw (2nd generation) – all cultivated in Denmark. The share of 2nd generation bioethanol exceeds 1st generation bioethanol towards 2030. Both scenarios initiate at a 5.75 % mixture in 2010 and reach 10 % and 25 % in 2030 for HS1 and HS2, such that the low mixture scenario reflects the Danish Act on sustainable biofuels (June 2009), implementing the EU renewable energy directive (2009/29/EC), using biofuels as energy carrier. The two scenarios are computed in two variants each, reflecting oil prices at 65$ and 100$ per barrel.

Background

Including analyses of the upstream energy use and environmental impacts of producing the biomass to be converted to biofuels is not yet common, while the need to do so has been highlighted in several studies, due to the impact it may have on the overall evaluation of the goods and bads of biofuel

(Searchinger, 2008). E.g. the details of how and where crops are grown, transportation and processing of crops into fuel are important in determining the net effect on greenhouse gas emissions (Horwath and Bringezu, 2008). These points have been approached in Well-to-Wheel analyses (e.g. J-E-C, 2007) for assessment of the impacts of biofuel production, while attempts to bring energy, emission and welfare economic costs together in the same consistent analytic framework is seldom found (Menichetti E., and M. Otto. 2009).

Data

Agricultural data expresses the specific production relations in Denmark and is established from various Danish sources, mainly the Danish Agricultural Advisory Service, both literature and personal communication, and represents probably the most coherent data on energy consumption in the agricultural sector, including upstream energy use. In general energy data stem from the Danish Energy Agency, which are mostly corresponding to international data. Other data, especially on process input products (fertilizer, pesticides, methanol etc) stems from the European WTW study (J-E-C, 2007), which again draws on central European data. Data on the biofuel manufacturing plants stems from J-E-C (ibid) regarding 1st generation bioethanol, Danish studies on 2nd generation bioethanol (Hedegaard Jensen and Thyø, 2007) and personal communication from a specific Danish plant regarding RME.

Method

The study analyses the consequences for society of realizing biofuel scenarios as compared to a reference situation with no biofuel. The method is taking its origin in the welfare economic analysis method as developed and reported in Møller and Slentø (2010). The integrated WTW method focuses on welfare economic changes to society and includes separate, however parallel running, WTW-analyses on energy consumption and emission changes assigned monetary values. It can be perceived as three individual analyses linked together. Or it can also be viewed as a welfare economic analysis subordinating and highlighting energy and emission analyses.

The welfare economic analysis method is characterized by the scarce resource approach, assuming that launching a new activity (e.g. biofuel conversion plant) removes resources from other activities that are substituted (e.g. fossil fuel refinery). Therefore abandoned activities must be included in the calculations, when analyzing cost and benefits to the society. By monetarizing as many different parameters as possible, both environmental externalities, production inputs, and marketed goods and services, the welfare economic approach is to be regarded as a multicriterial approach, though limited as other multicriterial approaches; limited, because not all decision aspects are possible to monetarize properly and meaningfully.

Basically, the consequential WTW analysis subtracts the net-benefits of the biofuel scenario from the corresponding net-benefits of the reference scenario, thus establishing the consequences of introducing the biofuel scenario for the society. Taking RME as example, in the reference situation, a crop (here wheat) is cultivated for food or feed, and fossil transport fuel is refined at the refinery. Those activities consume energy, emit pollutants and greenhouse gasses, and are associated with welfare economic costs and benefits.

In the scenario situation the former activities are abandoned and new ones emerge. Rape is cultivated on the former wheat area, the fossil fuel refinery is assumed closed down, and a new biofuel refinery is constructed. The new productions use various inputs, such as energy, fertilizers, chemicals etc. And next in order, those input products also consume energy in the production process. The side products of the biofuel production process (straw, rape seed cake etc.) will substitute similar goods at the marked, which consequently also belongs to the reference situation.

The method corresponds to the consequential LCA analysis method (ISO 14040-14044), which is an enhanced LCA analysis method where the narrow production path still is in the focus, but the analysis system is expanded to include the products on the marked that the side products substitutes

System delimitation

The analyzed system is delimited to the Danish border regarding emissions. In this way the calculated emissions consequences are comparable to official emission inventories of Denmark submitted to the international authorities (UN and EU). This implies that production emissions of imported products are not included. On the contrary energy consumption and economic calculations are not delimited. Prices are in terms of international market prices, which reflect production costs whether in Denmark or abroad. Consequently also upstream energy i.e. input product production energy, is included whether produced in Denmark or abroad.

A series of assumptions has been taken in the reference scenario in order to match real world, and also the biofuel scenarios are characterized by certain assumptions determinant for the results. Rape fields are assumed replacing wheat fields. Cereals and straw for 1st and 2nd generation biofuel stems from present wheat fields. 25 % straw from the fields in the reference situation is combusted directly in power plants, while the remaining part is either ploughed down or used as bedding or feed. In the scenario situation 100 % rape straw is assumed combusted directly in power plants. Moreover, regarding side products, rape seed cake and DDGS are substituting imported soy bean meal, molasses is substituting wheat feed, and glycerin and “dry matter” biomass is substituting imported coal as power plant fuel.

Policy scenario results

The first section of the table 1 beneath shows the main results of the Policy scenario (HS1) for 2010, indicating the percent changes shifting from reference situation with no biofuel, to scenario situation with oil prices at either 65$ or 100$. The results of the HS1-scenario are identical to the High mixture scenario (HS2) in 2010 because of equal mixtures at 5.75 %. Projection results 2010-2030 are omitted from this paper; however, typically, positive or negative trends in 2010 are continued over the years.

Policy scenario - 2010

65$-variant	RME	1.G. Biofuel	2.G Biofuel	***Total***
Energy consumption	-3%	7%	6%	***1%***
CO_2eq-emissions reduction	-65%	-46%	-33%	***-59%***
Welfare economic benefit	-40%	-32%	41%	***-33%***
100$-variant				
Energy consumption	-3%	7%	6%	***0,5%***
CO_2eq-emissions reduction	-65%	-46%	-33%	***-59%***
Welfare economic benefit	14%	13%	64%	***16%***

Table 1: Results of Policy scenario (HS1)

Regarding energy, the overall consequences of realizing the scenarios will be increased energy consumption at 1 %, however, at a disaggregated level, RME shows 3 % saving while 1st and 2nd generation bioethanol show 6 %-7 % increase in energy consumption. Regarding emissions, CO2 equivalent emissions (CO_2eq) are reduced at fossil emission displacement ratios at 65 %, 46 % and 33 % respectively for RME, 1st and 2nd generation biofuel, which are not directly comparable to other studies. Other air pollutants, not shown in the table, increase (NMVOC, CO, NO_x), decrease (particles -PM) or remain at the same level (SO_2, NH_3). Those changes are marginal in comparison to the overall emission levels in Denmark. Finally the results of the welfare economic analysis shows that with oil prices at 65$ per barrel, only 2nd generation bioethanol show benefits, while all three biofuel types show welfare economic benefits when the oil price exceeds 100$ per barrel.

Sensitivity analysis

A number of sensitivity analyses are carried out by producing variants of the scenario conditions – each changing one of the apparently important parameters. Due to limited space, only one variant “75 % straw” has been chosen to illustrate the high uncertainty in the results caused by scenario assumptions, see table 2. The sensitivity analysis represents a situation where 75 % of the straw in the reference situation is used as biomass fuel directly in power plants. This is in contrast to the basic reference where only 25 % in accordance with reality, is used for direct fuel, and where the remaining 50 % straw is mainly ploughed down and perceived as a free resource - despite its role in the soil carbon cycle.

The energy consumption changes are only affected little and remain at 1 % in total. The total CO_2eq-emission reductions drop from 62 % to only 25 % reductions in total. This, points at the possible disadvantage of producing bioethanol from straw in comparison to direct firing in power plants. The main reason is that the straw, which in the reference situation is used directly in power plants, has to be replaced with fossil coal when the straw now is used to produce bioethanol. Also the welfare economic costs and benefits change substantially. In the 65$ oil price variant, the total loss increases from 33 % to 55 %, and in the 100$ variant, the gains drops from 16 % to only 2 %.

"75 % straw" variant - 2010

65$-variant	RME	1.G. Biofuel	2.G Biofuel	***Total***
Energy consumption	-1%	6%	6%	***1%***
CO_2eq-emissions reduction	-26%	-46%	45%	***-24%***
Welfare economic benefit	-69%	-32%	-29%	***-55%***
100$-variant				
Energy consumption	-1%	6%	6%	***1%***
CO_2eq-emissions reduction	-26%	-46%	45%	***-24%***
Welfare economic benefit	-5%	13%	20%	***2%***

Table 2: Results of sensitivity analysis, "75% straw"

In detail, focusing on 2.G biofuel 100$ variant, there is an welfare economic benefit at 20 % introducing 2.G biofuel, as compared top reference situation, while CO_2eq emission *increases* at 45 % and energy consumption increases at 6 %. This shows the usefulness of the tree parallel running analyses on energy consumption, emissions and welfare economic changes; they are not necessarily concurrent in negative or positive consequences.

Discussion

Within the scenario assumptions, both the low and high mixture biofuel scenarios are socially feasible to realize when oil prices exceed and stay above 100$. RME would show positive balances both regarding energy consumption, CO_2 emissions and welfare economic benefits. However, giving up cereal production at the expense of rape seed, leads to an increased import of cereal feed, highlighting issues about security of local supply of food and feed in developing countries and further cut back of endangered forest. In the ethical perspective, and in general, the utilization of crop residuals like straw seems more promising; the technology converting straw into 2nd generation bioethanol is still immature and inefficient, leading to increased energy consumption, however welfare economically beneficial for the society. 1st generation bioethanol is a mature technology and welfare economically beneficial at oil prices above 100$ pr. barrel, however also energy inefficient, and the usage of wheat grains for the fermentation process leads to some extend to increased import of feed.

The strength of the integrated WTW-consequential analysis method is reflected in the different cost and benefit patterns of the energy, emission and welfare economic analyses, where e.g. welfare economic net-benefits are not necessarily associated with energy consumption or emission net-benefits, as seen in the sensitivity analysis “75 % straw”. The analyses are associated with uncertainty and depend on the actual scenario definitions and assumptions. The weakness of stand alone energy and emission analyses is their limited focus, which however can also be their strength, ensuring a consistent analysis framework. The strength of the welfare economic analysis is the integration of energy, emissions and other production factors by assigning them monetary values, while the weakness is the dependency on the actual availability and choice of prices.

According to the Danish government notice on sustainability criteria for biofuels (Danish Parliament, 2009), criterion for greenhouse gas reductions is minimum 35 % (increasing to 50% in 2017) per unit fossil fuel. The results of the present study cannot be compared directly to this criterion, since different in what is included in the calculations. The range of CO_2-replacement ratios in various studies are broad (UNEP, 2009) and dependent on the analytical framework chosen. Thought still not exhaustive, the present study has broadened the perspective on the feasibility of producing biofuels by conducting a broad consequential assessment and especially by detailed analyses of consequences in the agricultural sector.

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Emission scenarios for biofuel crop production in Denmark 2010-2030

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Introduction

Human enterprise has enormous impacts on the Earth's ecosystems, ranging from cellular changes in flora and wildlife in specific locales to changes in global flows of matter. Recent attention has focused on anthropogenic changes in the atmospheric carbon concentration, affecting global warming - a phenomenon potentially having an impact on most terrestrial live forms. Huge efforts have therefore been put into monitoring, analysing and understanding the global carbon balance, and not least methods to mitigate global warming. Among the methods is a transition from combustion of fossil fuels to increased reliance on renewable energy sources. Although critical voices have raised concern about reliance on fossil fuels for more than 40 years, only recently have we witnessed large-scale appraisal of liquid fuels based on biomass (biofuels). Theoretically capable of offsetting net carbon emissions due to the carbon sequestered during crop growth, and therefore environmentally benign, biofuel production have even earned it's own EU directive (2003/30/EC).

However, criticisms of biofuel production have also emerged. Some biofuel productions have shown not to be carbon neutral; during the biofuel production processes, as much, or more, carbon is emitted as the plants sequester during growth (Fargione *et al.*, 2008; Searchinger *et al.*, 2008). Currently, the EU sustainability criteria for biofuels specify that the carbon emitted during production of 1 MJ biofuel must not exceed 65% of the carbon emitted during production and combustion of 1 MJ fossil fuel. This threshold will change to 40% within 2018. Another concern with biofuel production is the land areas needed to produce the crops; the food/fuel dilemma has been widely discussed in public media, including the fact that demand for biofuels in the developed world puts pressure on both agricultural areas, wildlife areas and biodiversity hot spots. Albeit rather difficult to map and quantify precisely, such dynamic effects (or indirect land use changes, iLUC) have been studied (Wibe 2010), mostly focusing on cascading effects in the developing world.

An aspect of biofuel crop production not commonly discussed is the environmental impacts associated with crop production in input intensive agricultural systems (Menichetti & Otto, 2009; Simpson *et al.*, 2009). In Denmark, agriculture claims approximately 60% of the total terrestrial surface, with most of the arable land being cultivated intensively. Main crops are cereals and beets, cultivated as fodder to support a large livestock sector which in turn delivers manure to the cultivated fields.

The premise of this study has been that a shift from fodder production to biofuel crop production thus carries the consequence in Denmark of maintaining societal reliance on intensive crop production while reducing the amount of fodder produced, leading to decreasing livestock numbers and hence the amount of manure applied to the fields.

The primary aim of this paper has therefore been to estimate the changes in cropping intensities associated with a substitution of biofuel crops for fodder crops, measured as the amount of nitrogen, phosphorous and active pesticides applied and leached to fresh- and salt water bodies. Further, the speculated reduction in livestock numbers and subsequent manure stocks are likely to change the manure/inorganic fertilizer ratio, and thereby the emissions of ammonia to the atmosphere. Estimates of changes in the later were included in the impact assessments.

Methods

We constructed a transition matrix summarizing land use changes as envisioned by a biofuel production scenario for Danish agriculture (Larsen & Jepsen, in prep)). The scenario elaborates the assumption that the EU biofuels directive, demanding a 5.75% biofuel blend in the fuels consumed by the transport sector by 2010, and increasing to a 10% blend by 2020, is complied to, and further applies the normative assumption that the demand for biofuel crops are met by domestic supply.

The matrix summarizes changes in crop choices distributed on 30 crop types and changes in livestock numbers due to decreased fodder production. Next, based on coefficients used for the National GHG and emissions accounting and a review study of crop specific emission standards for N and P (table 1), we applied emission factors to the land use change matrix according to Equation 1.

$$Ex = \sum_{i}^{n} \sum_{j}^{n} LU_{ij} \cdot \left(Ex_j - Ex_i\right) \qquad \text{Equation 1}$$

Where *Ex* is the emissions of *x* (Nitrogen, Phosphorous or active ingredients in pesticides), *LUij* is the change in hectares from crop *i* to crop *j* and *(Ej-Ei)* is the emission changes of either N, P or application of pesticides associated with land use change from crop *i* to crop *j*.

Calculations of changes in ammonia emissions from manure is based on changes in the pig herd, the amount of manure produced by a standard pig, the nitrogen content of the manure (%) and an emission coefficient specifying NH3 emissions to the atmosphere per kg nitrogen in the manure (%) (Equation 2).

$$NH3_{manure} = \Delta pig \cdot (manure \cdot pig^{-1}) * N_{manure} \cdot C_{NH3pigs} \qquad \text{Equation 2}$$

Where $NH3_{manure}$ = ammonia emissions from pig manure, *Δpig* = the change in the pig herd, N_{manure} is the nitrogen content of pig manure and $C_{NH3pigs}$ is the NH3 emission coefficient for pig manure. The reduction in manure production results in an N deficit, seen from a farmer's perspective.

To compensate for the N loss, inorganic N fertilizer was assumed to replace the manure lost. Due to the different volativity of manure-derived organic N and inorganic N fertilizer, the ammonia emissions vary between the two N sources. We added the NH3 emitted from inorganic N fertilizer, $NH3_{fertilizer}$, by replacing the manure-specific NH3 emission coefficient of Equation 2 with a coefficient specific to inorganic fertilizer, $C_{NH3fertilizer}$ (Equation 3).

$$NH3_{fertilizer} = \Delta pig \cdot (manure \cdot pig^{-1}) * N_{manure} \cdot C_{NH3\,fertilizer} \quad \text{Equation 3}$$

Results

Table 1 Emissions to the aquatic environment (source: Mikkelsen 2010)

	N leached (kg/ha)	P surplus (kg/ha)	Pesticides, active ingredients
Cereals, spring-	65.27	-2	0.51
Rape, winter-	77.37	1	0.23

Table 2 Emission coefficients used for Equations 2 and 3.

	Organic manure	Inorganic fertilizer
Production per head per year	500 kg[a]	
Kg N / kg manure	0.06[a]	
Kg NH3/kg N input	0.149[b]	0.0225[b]
Kg N2O/kg N input	0.195[b]	0.195[b]

a) Dansk Landbrugsrådgivning (2010)
b) Slentø et al. (In press)

Direct land use changes caused by compliance with the EU biofuels directive in Denmark are not substantial; the largest land conversions by 2030 occur from fodder production of wheat and barley to rape production for biodiesel. Since our scenario build on the assumption that the diesel share of the total transport fuel composition will increase and due to the relatively low yields of rape, a total of approximately 200,000 ha cultivated with wheat and barley in the baseline will be converted to rape production in the scenario in order to meet the demand for biodiesel. All three crops are presently intensively cultivated. Further, some 100,000 ha of miscellaneous crops will be converted to rape production for biodiesel. The total emission changes related to introduction of domestic biofuel crop production calculated using Equation 1 are -467 tonnes P and 5,720 tonnes N. In addition, due to the relatively low pesticide application on rape, total active ingredient pesticide sees a decrease of -215 tonnes.

The reduced area with wheat decreases the amount of fodder available for pig production, yielding a decrease on the pig herd of around 2,800,000 heads and a resulting decrease in manure production of 1,400,000 tonnes. Equation 2 estimates the NH3 emission changes from manure at -12,524.4 tonnes, and the related change in CH3 emission from inorganic fertilizer is 1,417.5 tonnes (Equation 3), resulting in a net change of ammonia emissions of -11,106.9 tonnes.

Conclusion

Changes in crop choices and farming strategies induced by the EU biofuels directive have cascading effects, also within individual member states.

One such effect is an increase in the area used for biofuel crop production and subsequent decrease in area for animal fodder production. This implies a decrease in livestock numbers, and hence in the ammonia emissions associated with livestock production. Using land for biofuel crop production therefore seems a more environmentally benign practise in terms of atmospheric emissions than using area for animal production, due to the waste flows associated with current animal production practises. However, nitrogen emissions to the aquatic environment would increase substantially within this scenario, thereby bringing Denmark in collision with the EU water framework directive.

A more widespread utilization of the energy contained in the manure, eg. through production of biogas, will not only offset combustion of fossil fuels; it will also make nutrients contained in the manure more readily accessible to plants, and thus hamper nutrient leaching and emissions. While still not commonly practised, the use of biogasification of manure could potentially benefit both the environment and the rentability of livestock production.

An alternative way of mitigating eutrophication caused by intensive farming systems is by bioremediation. Particularly the use of willow to capture leached nutrients along streams and water bodies could potentially provide both biomass for energy purposes and cleaner agricultural production. The total efficiency of such initiatives yet remains unknown.

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External costs of conventional air pollutants for carriers of road transport

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Fossil fuels are used as input for as well power and heat generation as for transport services. In this presentation we explore the external costs of air pollution for the road transport sector which has been modelled in EVA-CEEH. Specific figures for the external costs from air pollution of transport are useful for energy system optimization that takes into account the extent of transport services required.

Modelling with the EVA-model has provided overall figures for the external costs of air pollution from the road transport sector, which represent an average for Denmark (Brandt et. al., 2009). The external costs are found to be significantly higher than for high stack point sources, due to the proximity of sources to the exposed individuals.

	SO_2/ SO_4	**NO_x**	**$PM_{2.5}$**
High-stack sources - emissions - total cost - unit cost/kg	13.760 tonnes 915 million 67 DKK/kg	48.826 tonnes 2093 million 43 DKK/kg	720 tonnes 57 million 80 DKK/kg
Transport sources - emissions - total cost - unit cost/kg	400 tonnes 95 million 239 DKK/kg	74.520 tonnes 4998 million 67 DKK/kg	2.120 tonnes 456 million 215 DKK/kg

Table 1: External costs from large power plants and the transport sector in Denmark according to EVA-CEEH (Brandt et.al. 2009; Andersen, 2010).

Exposure from road transport differs to some extent between urban and rural areas, mainly due to the differences in population densities in the local scale. While for the regional scale pollutants we do not find differences in the external costs according to the point of origin, the local scale exposure is important for the overall results when it comes to $PM_{2.5}$ and primary emissions of SO_2.

We use results from a high population density city exposure scenario achieved with an Urban Background Model (UBM) (Jensen, 2005) to derive correction coefficients for road transport emissions of $PM_{2.5}$ and SO_2 in urban areas with different population densities. We arrive at the results in table 2.

DKK/kg PM2.5	LOCAL SCALE	REGIONAL SCALE	SUM
Danmark (128 pers/km^2)	25	190	215
City class 1 (TU5, TU6) (1492 inh./km2)	297	190	487
City class 2 (TU4) (2392 inh./km2)	476	190	666
City class 3 (TU4) (2404 inh./km2)	479	190	669
City class 4 (TU4) (2547 inh./km2)	507	190	697
City class 5 (TU4) (2874 inh./km2)	572	190	762
City class 6 (TU4) (2792 inh./km2)	556	190	746
City class 7 (TU3) (2627 inh./km2)	523	190	713
City class 8 (TU3) (3494 inh./km2)	696	190	886
City class 9 (TU1) (6325 inh./km2)	1260	190	1450
Rural (TU7) (18 pers/km^2)	3	190	193

Table 2: Attribution of external costs to local and regional scale for different city class categories.

On basis of the COPERT database on air pollutant emissions, we further disaggregate the external costs to the various transport carriers for the current fleet of vehicles (Winther, 2006). As the emission profiles are specified according to the driving patterns it is possible to provide estimates for various road carriers according to road types. The availability of such figures allow road pricing schemes to include and reflect air pollution costs as part of more encompassing schemes to internalise external costs in market transactions.

TU1[5]: DKK/km	PM2.5	NOx	SO2	SUM
Private vehicle			0,01	0,09
- petrol	0,03	0,05	-	0,25
- diesel	0,20	0,05	0,02	1,31
Bus (diesel)	0,66	0,63	0,01	0,14
Van (petrol)	0,05	0,08	0,01	0,41
Van (diesel)	0,32	0,09	0,02	1,09
Truck (diesel)	0,62	0,45		

Table 3. External costs of air pollution from road transport in TU1 (Copenhagen).

TU4[6]: DKK/km	PM2.5	NOx	SO2	SUM
Private vehicle			-	0,06
- petrol	0,01	0,05	-	0,10
- diesel	0,05	0,05	0,01	0,80
Bus (diesel)	0,16	0,63	-	0,10
Van (petrol)	0,01	0,08	-	0,17
Van (diesel)	0,08	0,09	0,01	0,61
Truck (diesel)	0,15	0,45		

Table 4. External costs of air pollution from road transport in TU4 (City > 10.000).

[5] TU1 is municipality of Copenhagen.
[6] TU4: 10.000-100.000 inhabitants.

TU7[7]: DKK/km	PM2.5	NOx	SO2	SUM
Private vehicle			-	0,04
- petrol	-	0,04	-	0,04
- diesel	0,01	0,03	0,01	0,52
Bus (diesel)	0,06	0,46	-	0,08
Van (petrol)	0,01	0,07	-	0,08
Van (diesel)	0,03	0,06	0,01	0,43
Truck (diesel)	0,06	0,37		

Table 5. External costs of air pollution from road transport in TU7 (rural).

HIGHWAY DKK/km TU1	PM2.5	NOx	SO2	SUM
Private vehicle				
- petrol	0,02	0,06	-	0,08
- diesel	0,13	0,04	-	0,17
Bus (diesel)	0,37	0,39	0,01	0,77
Van (petrol)	0,03	0,08	-	0,12
Van (diesel)	0,22	0,06	-	0,28
Truck (diesel)	0,38	0,33	0,02	0,73

Table 6 Highway: External costs of air pollution from road transport in TU.

HIGHWAY DKK/km TU4	PM2.5	NOx	SO2	SUM
Private vehicle				
- petrol	-	0,06	-	0,06
- diesel	0,03	0,04	-	0,07
Bus (diesel)	0,09	0,39	0,01	0,48
Van (petrol)	0,01	0,08	-	0,09
Van (diesel)	0,05	0,06	-	0,11
Truck (diesel)	0,09	0,33	0,01	0,43

Table 7 Highway: External costs of air pollution from road transport in TU4.

[7] TU7 are rural areas without towns and villages.

HIGHWAY DKK/km TU7	PM2.5	NOx	SO2	SUM
Private vehicle				
- petrol	-	0,06	-	0,06
- diesel	0,02	0,04	-	0,06
Bus (diesel)	0,05	0,39	-	0,44
Van (petrol)	-	0,08	-	0,08
Van (diesel)	0,03	0,06	-	0,09
Truck (diesel)	0,05	0,33	0,01	0,38

Table 8. Highway: External costs of air pollution from road transport in TU7..

As shown in the table the highest external costs for air pollution are unsurprisingly found for buses and trucks using diesel as propellant. However, the lowest costs are somewhat less intuitively also accorded to diesel driven vehicles, i.e. private diesel cars driving in rural areas. NOx emissions are lower for the historical fleet of private diesel cars than for the fleet of petrol cars. However, among private cars adhering to present EURO-norms the NOx-emissions per kilometre for diesel cars exceed those for petrol, hence providing the expected result for the external costs. Some implications: 1) If in average two persons are transported in a petrol car, a diesel bus needs about 25-30 passengers before there is a break-even between the external air pollution costs per person (excl. CO_2). 2) In order to match the external costs for a private diesel car without a diesel filter, the annual penalty fee for 17,000 km of driving should amount to 1700-4300 DKK/year in cities and 700 DKK/year in rural areas.

DKK_{2006} per. km	Rural	City (10.000 inh.)	Urban (CPH)
Private vehicle			
- petrol	0,04	0,06	0,09
- diesel	0,04	0,10	0,25
Bus (diesel)	0,52	0,80	1,31
Van (petrol)	0,08	0,10	0,14
Van (diesel)	0,08	0,17	0,41
Truck (diesel)	0,43	0,61	1,09
Weighted average	0,08	0,12	0,22

Table 9: External costs of air pollution from road transport.

DKK_{2006} per. Km HIGHWAY	Rural	City (10.000 inh.)	Urban (CPH)
Private vehicle			
- petrol	0,06	0,06	0,09
- diesel	0,06	0,07	0,17
Bus (diesel)	0,44	0,48	0,77
Van (petrol)	0,08	0,09	0,12
Van (diesel)	0,09	0,11	0,28
Truck (diesel)	0,38	0,43	0,73
Weighted average	0,09	0,10	0,16

Table 10: External costs of air pollution from road transport - highways.

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Assessment of Health-Cost Externalities of Air Pollution at the National Level using the EVA Model System

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Introduction

Air pollution has significant negative impacts on human health and well-being, which entail substantial economic consequences. We have developed an integrated model system, EVA (External Valuation of Air pollution; Frohn et al., 2005; Andersen et al., 2007; Brandt et al., 2010), to assess health-related economic externalities of air pollution resulting from specific emission sources or sectors. The EVA system was initially developed to assess externalities from power production, but in this study it is extended to evaluate costs at the national level from all major emission sectors. In this study, we estimate the impacts and total externality costs from the main emission sectors in Denmark, representing the 10 major SNAP codes. Furthermore, we assess the impacts and externality costs of all emissions simultaneously from the whole of Europe as well as from international ship traffic in general, since this sector seems to be very important but is currently unregulated.

The EVA Model System

The basic concept of the EVA system is based on the impact pathway chain (see Fig. 1) and consists of a regional scale Eulerian atmospheric chemistry-transport model, the Danish Eulerian Hemispheric Model (DEHM), (Christensen 1997), including detailed emissions inventories, address-level or gridded population data, state-of-the-art exposure-response functions and monetary valuation of the impacts from air pollution applicable for European conditions.

The essential idea behind the EVA system is to develop and apply state-of-the-art methodologies in all individual links of the impacts pathway chain. Other comparable systems commonly estimate the non-linear behaviour of atmospheric chemistry and deposition processes using linear approximations. The EVA system has the advantage that it describes such processes using a comprehensive, state-of-the-art chemical transport model when calculating how specific changes to emissions affect air pollution levels. The model covers the northern hemisphere to describe the intercontinental contributions and includes higher resolution nesting over Europe (see Fig. 2). All scenarios are run individually and not estimated using linear extra-/interpolation from standard reductions.

However, quantifying the contribution from specific emission sources to the atmospheric concentrations levels is a challenge, especially if the emissions of interest are relatively small. Numerical noise in atmospheric models can be of a similar order of magnitude as the signal from the emissions of interest. Therefore, we developed a new “tagging” method (see Fig. 3), to examine how specific emission sources influence air pollution levels, without assuming linearity of the non-linear behaviour of atmospheric chemistry and diminishing the influence from the numerical noise. This method is far more precise than taking the difference between two concentration fields.

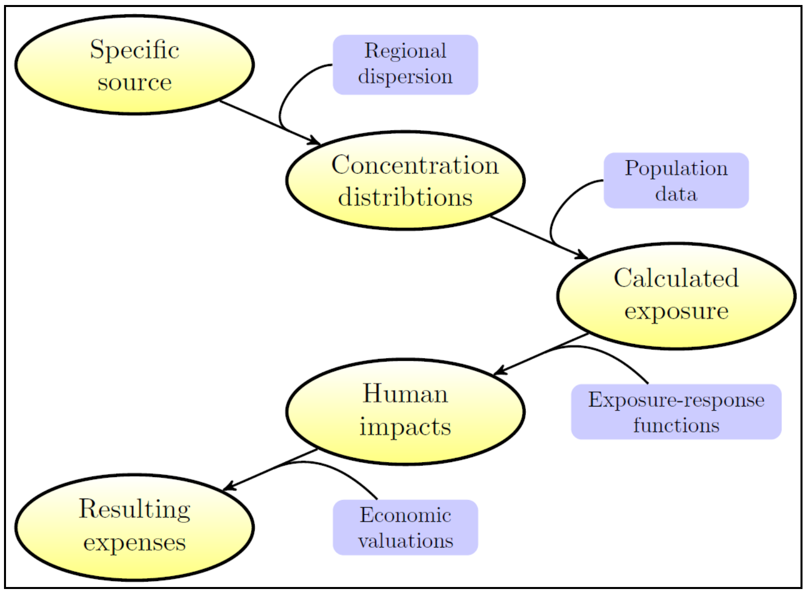

Figure 1: A schematic diagram of the impact-pathway methodology from site-specific emissions, atmospheric transport and chemistry, human exposure estimated using population data, to the resulting human health impacts and related costs.

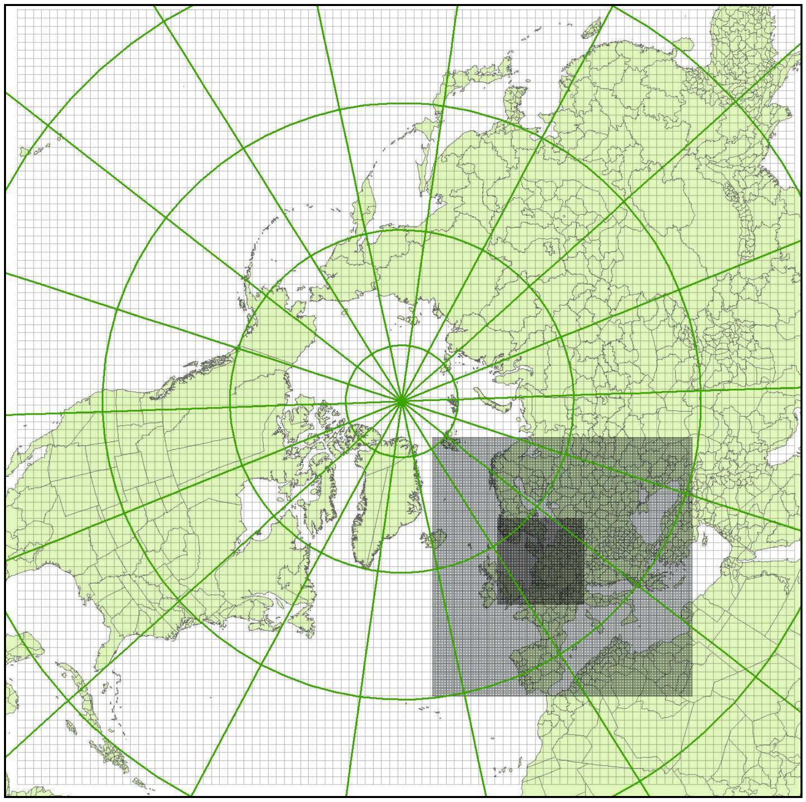

Figure 2: The DEHM domain (polar stereographic projection) with three domains. The mother domain covers the northern hemisphere with a resolution of 150 km x 150 km. The two nested domains included have resolution of 50 km x 50 km and 16.67 km x 16.67 km, respectively.

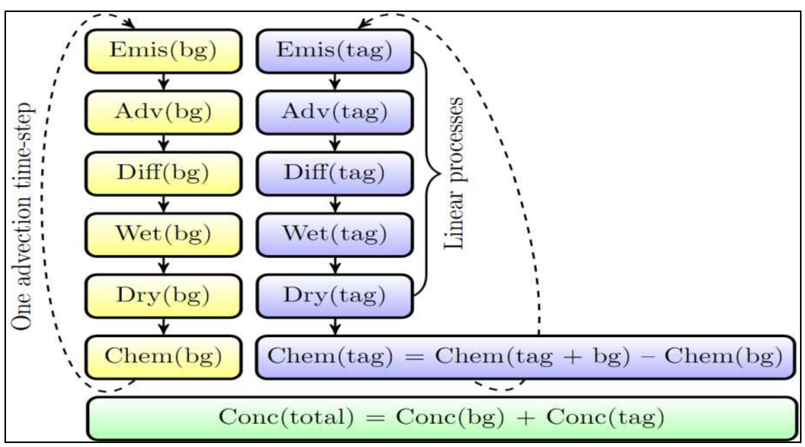

Figure 3: An overview of the tagging method. The concentration fields for a specific emissions source (tag) is modelled in parallel with the background field (bg) in the CTM. The need for tagging is due to the non-linear processes: atmospheric chemistry (Chem). The linear processes are emissions (Emis), advection (Adv), atmospheric diffusion (Diff), wet deposition (Wet) and dry deposition (Dry). For the non-linear processes, the tagged concentration fields are estimated by first adding the background and tag concentration fields, then applying the non-linear operator (e.g. the chemistry). The concentration

field obtained by applying the non-linear operator to the background field alone is subtracted. Thus the contribution from the specific emissions source is accounted for appropriately.

All species with exposure-response functions given in literature are included in the study. For compounds in aerosol phase, the impacts are assumed to be proportional to their contribution to the particle mass. Presently, the species included in the EVA system are: O_3, CO, SO_2, SO_4^{2-}, NO^{3-}, and the primary part of $PM_{2.5}$.

Results and conclusions

In table 1, the total health-cost externalities for Europe and Denmark from the different emission sectors are given. The SNAP codes are: 1) Combustion in energy and transformation industries, 2) Non-industrial combustion plants, 3) Combustion in manufacturing industry, 4) Production processes, 5) Extraction and distribution of fossil fuels and geothermal energy, 6) Solvents and other product use, 7) Road transport, 8) Other mobile sources and machinery, 9) Waste treatment and disposal, 10) Agriculture, 15) International ship traffic.

The basic questions that we wanted to test with the EVA system are: 1) What are the relative contributions to impacts on human health and related externality costs from the 10 major emission sectors in Denmark? 2) What are the total impacts due to all the emissions in Denmark? 3) What are the health impacts and cost from international ship traffic over Europe and Denmark? 4) What are the total impacts and related costs from all emissions in the northern hemisphere in Europe and in Denmark? All results covering the four questions are given as impacts/cost for the whole of Europe and similar inside Denmark itself – the latter being part of the first. For the international ship traffic and for all emissions from the whole of Europe, results for four different years were made (2000, 2007, 2011 and 2020) in order to examine the evolution over time. For the years 2000, 2007 emissions from the EMEP database were used. For the year 2020, the total emissions for each country are based on the National Emission Ceilings version 2 (NEC-II) but using the 2007 emission distribution. The 2011 emissions are based on the 2007 emissions but with changes according to the agreements within the Sulphur Emission Control Areas (SECA) for international ship traffic, where the sulphur contents the heavy fuel are reduced from 2.7% in the year 2000 to 0.1% in the year 2020. The SECA areas include the North Sea and the Baltic Sea.

Sector	**Emission year**	**Europe** **Mio Euros**	**DK** **Mio Euros**
DK SNAP 1: Major power plants	2000	527	38

DK SNAP 2: Smaller power plants	2000	462	138
DK SNAP 3: Manufacturing industry	2000	247	23
DK SNAP 4: Production processes	2000	47	11
DK SNAP 5: Extraction/distribution of fossil fuels	2000	40	2
DK SNAP 6: Solvents and other product use	2000	120	4
DK SNAP 7: Road transport	2000	979	193
DK SNAP 8: Other mobile sources	2000	437	75
DK SNAP 9: Waste treatment and disposal	2000	37	1
DK SNAP 10: Agriculture	2000	2648	409
DK-sum 1-10: Sum of all of the above	2000	5543	893
DK-all (all SNAP): All DK sectors simultaneously	2000	5192	843
Europe SNAP 15: International ship traffic	2000	56351	802
Europe SNAP 15: International ship traffic	2007	54735	621
Europe SNAP 15: International ship traffic	2011	52186	557
Europe SNAP 15: International ship traffic	2020	61248	483
Europe-all: All emissions from Europe	2000	781340	4529
Europe-all: All emissions from Europe	2007	661220	3790
Europe-all: All emissions from Europe	2011	657720	3742
Europe-all: All emissions from Europe	2020	519420	2529

Table 1: Total health-cost externalities for Europe and Denmark for the 10 major individual emission SNAP categories for Denmark (DK SNAP 1-10) and their sum, for all emissions in Denmark (DK-all), for international ship traffic (Europe SNAP 15), and for all emission in the whole of Europe (Europe-all). For the latter two categories, the calculations were carried out for four different emission years. All costs are in 2006 prices.

The results in table 1 show, for example, that the total health cost in Europe from all Danish emissions is estimated to 5192 Mio Euros/year, while the same emissions account for the cost of 843 Mio Euros/year in Denmark alone. The relative contributions from the major SNAP categories in Denmark show that the agriculture, road traffic and the power production are the major contributors. The external cost from the international ship traffic is more than 50 bn. Euros/year. The valuation of the total air pollution levels in the whole of Europe is estimated to be around 780 bn. Euros/year in the year 2000 and around 519 bn. Euros/year in the year 2020. Our estimates of the external costs of total air pollution levels in Europe are similar to results presented in the last baseline report from CAFE 2005 (Watkiss et al., 2005). For example, according to CAFE: ”Core estimates of annual health damage due to air pollution in 2000 and in 2020 in EU25 between 276 - 790 bn Euro”, which are close to the EVA results. In table 2, the externality costs for each species have been divided by the emissions for the specific runs, giving the cost per kg of the emission divided into species and sectors. The cost per kg is in the Centre for Energy, Environment and Health (CEEH) used as input to the energy optimisation model, Balmorel.

Sector	Emission year	CO [C]	S [S]	N [N]	NH_3 [N]	$PM_{2.5}$
DK SNAP 1	2000	0.001	22.2	23.9	-	19.2
DK SNAP 2	2000	0.002	32.7	33.8	-	28.4
DK SNAP 3	2000	0.001	27.0	27.6	-	19.8
DK SNAP 4	2000	0.014	44.9	110.5	-	41.2
DK SNAP 5	2000	-	-	-	-	26.2
DK SNAP 6	2000	-	-	-	-	-
DK SNAP 7	2000	0.003	188.5	33.3	-	44.4
DK SNAP 8	2000	0.002	22.9	28.6	-	34.1
DK SNAP 9	2000	0.000	20.0	38.9	-	-
DK SNAP 10	2000	-	-	-	30.7*	20.4
DK-all (all SNAP)	2000	0.002	102.5	54.5	-	31.2
Europe SNAP 15	2000	0.000	26.7	26.3	-	22.1
Europe SNAP 15	2007	0.000	23.5	25.9	-	18.9
Europe SNAP 15	2011	0.000	22.4	26.2	-	18.2
Europe SNAP 15	2020	0.000	20.8	26.4	-	17.0
Europe-all	2000	0.006	31.0	50.9	-	46.2
Europe-all	2007	0.006	29.7	48.6	-	47.3
Europe-all	2011	0.006	29.5	48.6	-	47.3
Europe-all	2020	0.006	27.7	51.3	-	44.0

Table 2: Cost per kg emission (Euros/kg -C -N) for the 10 major individual SNAP categories for Denmark (DK SNAP 1-10), for all emissions in Denmark (DK-all), for international ship traffic (Europe SNAP 15), and for all emission in the whole of Europe (Europe-all). For the latter two categories, the calculations were carried out for four different emission years. *The cost per kg related to NH_3 emissions is related to the dose-response of S and N in the agricultural sector, due to the chemical transformation of NH3 transforms into SO_4NH_4, $(SO_4)_2NH_4$ and NO_3NH_4.

Acknowledgement

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Session 5: Future Energy Scenarios in Integrated Assessment Models

Keynote speaker: Maryse Labriet, ENERIS, Spain

Chair: Pia Frederiksen

A transdisciplinary approach to the use of scenarios for energy production and consumption

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The transdisciplinary field 'Planning, Space and Resources' (PSR) is developed at our university and builds upon economics, political science and sociology. Courses in these disciplines were separated in the 1990's with PSR showing how to reintegrate them e.g. by incorporating space and time into theoretical reasoning (RCD1999). Suggested by the Gulbenkian Commission on Restructuring of the Social Sciences (Wallerstein et al.1996), an example of this endeavour is world-system analysis. It looks at 500 years of globalisation of the Euro-centric world-system as a contradictory process of expansion which now asymptotically meets its final limits (Wallerstein 2004). Today, it is said to have entered into a chaotic phase of transition with ever wider oscillations in core areas of economic, political and socio-ecological reproduction. This perspective upon societal reproduction frames the following remarks on the uses of scenarios as a methodological answer to the challenges of transition. Emerging real-world choices and bifurcations must be made better understandable, because the world-system is passing through them since several decades. But even such hopeful social and political actors as those who met last year in Copenhagen around COP 15 had apparently no common understanding of them, so that better ways have to be found, how to translate scientific evidence into politics (RCD 2010).

How to understand sustainable world-system development

The perspective of a globally shared future, originally projected by the World Commission on Environment and Development ("Our Common future", Brundtland 1987) was based upon three basic understandings of the very concept of sustainable development (SD): a normative, an analytical and a political (Becker and Jahn 1999). Correspondingly, the Brundtland Report itself contains three definitions of SD which together can help to avoid the misunderstanding of Herman Daly, who called it merely a 'pre-analytical vision' (Daly 1996).

(a) Daly reports namely only the *short, normative vision* of SD as an ideal of justice in the use of resources between generations of humans, meaning that the potentials to meet their basic needs are maintained. A critical look at the implied concepts of needs and aspirations reveals, however, that global inequalities make this vision ambiguous: it contains both basic needs of the World's poor and aspirations for a better life of the population of rich countries. This difference gives rise not only to different claims on ecosystem services, but also to environmental movements with different arguments, goals and strategies (Mjøset 2002).

(b) There is, however, also an *analytical definition* stating that SD as a process of harmonisation consists of different processess of change: resource exploitation, orientation of technological development, stabilisation of investments and institutional change (RCD 2002, 2003). The aggressive term *resource 'exploitation'* has since 1987 shown its problematic meaning: to exploit natural resources e.g. for energy purposes has become a prerogative of 'high politics' leaving the other components behind, even if they might be necessary for SD. When the aim of procuring cheap oil from the Middle East became dominant, the strategic energy alternative, delineated in chapter 7 of the 'Brundtland'-report as an alternative between low and high energy scenarios, was, thus, decided upon in favor of the latter. While the realization of a low energy path was known to demand profound *institutional changes* in the internal set-up of societal reproduction, the high energy road demanded only politically to undermine the collective security order of the United Nations. Seen from the point of view of anti-environmental and pro-privatisation movements, which still lie at the core of the enduring turmoil in Iraq (Hakes 2008, 100), public interference with the *direction of technological innovations* and with the *lack of continuity of investments* is not at all desirable.

(c) The concluding section of the sub-chapter of the Brundtland Report 'Towards SD' finally contains a *political conceptualisation of changes* at world-system level *necessary* to reach SD as e.g. a production system that respects the obligation of preserving the ecological foundations of development. This explains, why neoliberal globalisation and its accompanying revival of geopolitics became a barrier against the impetus of the Commission towards a harmonious world-wide development (RCD 2006). This happened, not the least, by 'postponing' the so-called post-Cold-War Peace Dividend and its possible uses for an ecological restructuring of global production that could have prevented global warming (Commoner 1990, RCD 2009b). The warnings of the 2006 Stern Review on the economics of climate change that costs rise progressively with waste of time in this regard, can, thus, retrospectively be applied at least from 1990 – and should be extended to comprise broader categories of costs as those from warfare (RCD 2003, 2006, 2010).

Scenario methodology articulates intentions of decision-making with real-world premises

The inflationary use of the term 'scenario' cannot conceal its misuse in case of mere variants of otherwise homogeneous projections (Robinson 2003). Already in 1974 Mesarovic and Pestel gave an explicit description of the merits of the scenario method in their 2nd report to the Club of Rome. It combined a multi-layered causal model with a normative-institutional approach to decision-making. In this way, scenario methodology is apt to furnish a consistent analytical underpinning to complex issues of how to reach SD (RCD 2003).

The multi-polar regionalization of the Mesarovic-Pestel model, which was its distinction from the First Report to the Club of Rome focussing upon global physical limits to growth (Meadows and Meadows 1972), could, however, not be translated into planning reality. Its intention had been to help mediate interest conflicts between oil producing and oil consuming countries in a world-wide bargaining economy. This intention was, however, frustrated in real history. Something was wrong with the premises of this optimistic model itself: The emerging transnational power elites preferred to scape-goate, split and make Arab-OPEC countries the object of manifold military interventions.

By presenting alternative energy scenarios, intellectuals within the West-German green movement succeeded, however, early to transport into discursive space the methodology to reach a turning point in energy politics away from nuclear and oil and towards renewable energy as well as rational energy use. Already in 1979, the former unseen inverse-U-shape of (low) energy demand curves emerged in parliamentary hearings with a promise of decline coupled with sufficient energy services in end-use (Eco-Institute, Energiewende 1980, RCD 2009a). After the era of chancellor Kohl, when a red-green government belatedly came into power in 1998, this impetus led to plans of out-phasing nuclear power in Germany.- As the spectre of an 'energy gap' now is projected on the global level, this result of an enlightened public discussion is worth reminding, as it has falsified wrong deterministic assumptions and, thus, demonstrated the principal feasibility of more desirable, collectively agreed futures in the sense of Robinson 2003.

Also, the Danish example of energy politics since the mid-1970's is worth remembering, as it was situated at the cross-roads of Brundtland recommendations and EU liberalisation strategies. Already in 1989, some of my first year students articulated this in a newspaper feature article (Markussen et al.1989). In the beginning of the 1990's, alternative energy scenarios were, then, transformed into government policy. The official plan 'Energy 2000' was published with a special report on 'Brundtland scenarios', written by RISØ researchers. This was decisive for a successful restructuring of the electricity sector making it more flexible for decentralised and fluctuating inputs from renewable energy and co-generators of heat and electricity (RCD 2010).

Regarding the desirability of a better future and the means to achieve it, the pivot of scenario writing is back-casting: "The problem-driven nature of back-casting and the need to incorporate the values and preferences of different stakeholders...imply that back-casting models should be able to address concrete social problems and speak to non-expert users" (Robinson 2003, 845). It presupposes, thus, "to recognize the potential existence of multiple worlds" - an assumption which, however, is at odds with "much climate policy analysis"(Robinson 2001). This has "to date...been predicated upon the assumption of a single baseline for underlying socio-economic dynamics." The contradiction between this world-view saying "There Is No Alternative" (TINA) and the credo "Another World Is Possible" can by back-casting be resolved into support for the latter and help to integrate climate futures into strategies for sustainable development (Robinson and Herbert 2001).

Optimisation, conversion and substitution of resources as a planning challenge

Three kinds of innovations in materials and/or resource use can de distinguished, namely processes of optimisation, held within the limits of comparative statics; of socio-technical conversion in uses e.g. from military to civilian purposes; and full-fledged substitution of products or processes. The latter is an important alternative to end-of-pipe regulation which only gradually reduces – or: optimises - environmental pollution. Whereas this may imply high costs, lower efficiencies and with expanded use even rebound effects, substitution is more apt to real problem solving (Commoner, 1990, RCD, 2009b). Barry Commoner has stressed this by saying: "If You don't put something into the environment, it isn't there" (p.43). If tetra-ethyl lead is not added to motor vehicle fuel, it is not in the environment of roads. This substitution was a long time resisted by oil companies and motor vehicle producers. Today it has prevented more children from becoming restrained in their development because of brain damages. Why is it so difficult to transfer this socio-technological insight and the precautionary principle of environmental politics in general into planning processes, when they – rightly developed - could relieve our global environment from the spectre of climate change and living populations (human and non-human) from local pollution, while being development-friendly in the global field?

Firstly, there is the still omnipresent denunciation of a failed planned economy in the state socialist block that emerged after the First World War and expanded after the Second. In case of East Germany, one has to remember that the most toxic production complexes around the chemical cluster of Bitterfeld were derived from the decision of the German military commanders after the First World War in strategically secure surroundings to establish facilities for the conversion of brown coal to gasoline (RCD 1992). The political directory of the central planning system in the GDR was, however, specifically responsible for extending the use of brown coal until the last years of the GDR: Without alternative, they had only planned to substitute nuclear power for brown coal in energy procurement (RCD 2009a). In the wake of the explosive melt-down of the Chernobyl nuclear power plant in 1986 this path revealed itself to be a mission impossible. Although the Soviet Union in 1963 had been co-responsible for ending atmospheric nuclear tests, they were, however, under Cold War conditions exposed to geopolitical pressures to continue with nuclear technology. And when the hoped-for Peace Dividend after the Cold War by U.S.-American initiative was postponed in 1990, the chances to substitute clean renewables for nuclear power and thus to prevent global warming were equally diminished (RCD 2009b).

Secondly, elites of environmentally greening countries have promoted political decisions to out-phase fossil fuels. This seems principally to be a correct decision in order to cure the problem of man-made climate change through cumulative additions of CO2 to the atmosphere. Already COP 3 at Kyoto 1997 could, however, politically have ended as a failure, if countries had not accepted a last-minute proposal by Australia to include into the Kyoto-protocol the compensatory mechanism of CO2-binding by living vegetation (green, photosynthesising plants). In book-keeping on countries' CO2-records, this can, since then, offset continued emissions (Flannery 2006). In 1998 the German advisory council on environmental issues (WGBU) protested publicly against this confusion, but had to give in.

Andreas Fischlin of IPCC working group 2 has shown that a rise of 2,5 ^{o}C above pre-industrial levels probably will lead to forests becoming a net source of CO2 (see also Hakes 2008, 106). In connection to COP15, the International Union of Forest Research Organisations presented this finding at a special session of Forest Day 3 (IUFRO 2009). Here, a representative of the European wood industry rejected any idea of revising the profitable practice of giving CO2-neutrality credits for wood combustion. Sweden also propagates the dogma of zero-CO2-emissions from this misuse of resources (Swedish EPA, 2009). The non-energetic use of wood is, otherwise, far superior in mitigating climate change: Durable products manufactured from wood substitute for cement, steel or petrochemicals, which results in enduring emission reductions.

When the combustion of fuel wood substitutes for fossils, especially natural gas, the same cannot be said. Immediately, CO_2 emissions are twice as high or even more (see www.maforests.org/MFWCarb.pdf). In the middle run, additional growth of plant matter with long rotation times as trees produces a 'time lag' between emissions and binding of CO_2, which “may lead to climate alterations”(Sørensen 2004). It is therefore an irresponsible misuse of power that Danish authorities in their 2006 report to the Stockholm Convention on out-phasing persistent organic pollutants because of an alleged climate political advantage allowed further emissions of toxic substances as PAH and dioxins from small wood stoves (RCD 2009b). Besides landscape and, thus, food contamination direct inhalation exposure to dioxins leads to reductions in cerebral blood flow – a phenomenon not accounted for in risk assessments (Fabig 1998).

Local health-environment connections from wood combustion

Without interfering with the global CO_2-neutrality assumption for wood combustion, the Danish Association of Engineers (IDA) demonstrated considerable reductions of societal health costs, if private wood combustion is transferred to co-generators (IDA 2009). In order to reach realistic exposure assessments for neighbours of wood stoves, IDA applied an EVA/EECH model with a grid of 5 x 5 km and a sector-identification e.g. for households affected by PM 2,5 exposure.

Health effects of population exposure to particles in wood smoke have earlier been evaluated very insufficiently (Danish Ministry of the Environment, 2008): Instead of a grid combined with data on population densities, the Ministry used a model spreading the sum total of PM 2,5 emissions over the whole area of Denmark. Together with too low dose-response estimates taken from WHO and a selection of the lowest measured values from field tests, the report underestimated health damages and assumed only around 200 premature deaths – out of around 55 000 yearly cases. The critical exposure to dioxins was also systematically overlooked. Their concentration in the exhaust air of wood stoves has been shown to transgress the EU limit value for emissions from high, elevated smoke stacks. While *they* may assure free dilution, emissions from wood stoves enter into whirl zones near the ground with much turbulence.- It is really “time to put the human receptor into air pollution control policy”, as William W. Nazaroff recently said in a special contribution to Atmospheric Environment (Nazaroff 2008).

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Enabling and governing transitions to a low carbon society

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Introduction

Modern societies are facing the challenge of a transition towards low carbon energy systems. This challenge has been addressed by research within different disciplines and with very different approaches reaching from focusing on innovation in the energy supply systems and natural resources for the production of energy, to developing policy and market measures that can guide behavioral changes from consumers and companies, and to provide scenarios documenting the costs of not acting and potentials for restructuring society's energy system.

Most of the funded research in Denmark has hitherto had its focus on policy instruments, market mechanisms, innovations, costs and impacts as well as optimization within known options and impacts. The Danish Strategic Research council has funded a four year research project (a research alliance) with the title "Enabling and governing transitions to a low carbon society". The aim of this project is to conceptualize the dynamics of transition processes within society involving the diverse set of actors from consumers to companies and organizations. Instead of documenting that change is needed and that it is possible at least in principle this research will analyze how actors engage in changing behavior and how these processes might link up and result in a transition so much needed to reach the climate policy goals that may not result just from attempts to define top-down policies and economic measures. The research will focus on how actors engage in change processes and why instruments as e.g. incentives, regulation and institutions sometimes succeed and in other situations fail to facilitate dynamic transition (Jørgensen & Strunge, 2002).

Current attempts to coordinate and facilitate transition in Denmark indicate that a wider endorsement from actors at different levels of society is lacking in order to ensure the necessary momentum of the process.While contemporary research in relation to transitions in energy systems demonstrates the potential for transformations of the energy systems, it does at large not provide the tools to engage societal actors in implemting desired chaanges in order to achieve the transitions so much needed.

Theoretical foundation

Technological opportunities and visions do not automatically lead to implementation and change. The complexity of the required solutions makes technical and social innovations necessary elements in transforming the existing energy regimes (Freeman & Perez, 1988). Existing international research in systemic transition argue that actors at all levels of society need to become engaged in and contribute to solutions, since sustainable transitions involve innovations at multiple levels in society (Carlsson &

Stankiewitz, 1995; Vleuten & Raven, 2006). The research takes the outset in current research in patterns of systemic transition (Geels & Schot, 2007), but develops this research area further by also conducting studies of ongoing societal changes. Transition of the path-dependent, socio-technical regimes in the energy system is a governance challenge, since transitions need to occur simultaneously in different arenas without necessarily having a specific 'centre' of co-ordination. Changes of regimes require innovative breakthroughs in technology, changes of institutional frames and changes in social practices, but also increased utilisation of well known solutions is important. The overall strategy is to work with cross-cutting issues of transition at four key transition arenas in society: policy, households, companies and cities. Through an integrative combination of historical analysis, case studies and action research, the research project will analyse the roles of socio-technical experiments, creation and utilisation of 'windows of opportunity' and stabilisation of changes in societal niches into regime transformation. The project provides knowledge and methods to enable different stakeholders' development of governance strategies in order to handle and navigate through the complex and conflicting nature of transition processes.

Current research provides mainly a retrospective insight into dynamics of technology development, and these studies of former transitions will produce valuable knowledge and form the background for research on governance and measures that can respond to and utilise conflicts as constructive transition elements. The project builds upon current research in patterns of systemic transition (Geels 2004), but develops this research area further by also conducting studies of ongoing societal changes. Specifically, the research agenda will adress the following hypotheses:

- Technical opportunities do not become 'working solutions' until they are specified within an institutional and regulatory context.
- Dynamics of innovation and change is governed by a complexity of interpretations and conflicts that are part of the process of change.
- Transitions occur based on actors becoming aligned and interconnected while still maintaining different identities and interests within patterns of socio-technical experimentation.
- Changes in socio-technical regimes is linked to the emergence of 'windows of opportunity', where general discoursive movements can support the appearance of new configurations.
- Transition may be advanced by innovation activity but also by regime interaction or through the influence of overlying societal discourses.

Empirical focus

The project will engage with actors from five different societal arenas in which the project will study experiences with network based governance and characterise different configurations in socio-technical innovation processes. The arenas are:

- Interactions between market-based instruments, regulatory policy and voluntary certification
- Challenging regimes at the level of everyday life in household practises
- Dynamics of innovation in energy systems of production, use and savings
- Municipalities as intermediaries reconstructing relations between cityscape and mobility
- Governance of bioenergy: Trade, environment and integration of energy system

Energy scenarios can play an important role in establishing guiding visions (Schot & Geels, 2008). Especially when they support the involved actors' motivations and at the same time provides goals and measures that support action. The project is not expected to develop new energy scenarios, but integrate (through dialogeue with the relevant projects) the contemporary research, which offers valuable background and input regarding technical innovations. This goes particularly for projects including economic models and scenarios for such transformations of the composition of energy supplies with some emphasis on energy services and changes in behaviour of energy users: Coherent energy and environmental system analysis (CEESA), Renewable Energy in the transport sector using Biofuels ans Energy Carriers (REBECa), Center for Energy Environment and Health (CEEH).

These current projects are developing output with regards to environmental, social, economic and health aspects of future energy scenarios. It will be an important element in the transitions project to study how these results are integrated and mobilized in different types of governance processes. Knowledge from environmental impact modelling studies may be used to confirm existing institutional paradigms but can also function as the starting point of new agendas and socio-technical experiments.

Objectives and outcomes

The main objective of the project is to develop knowledge and strategies to enable a stronger and more coordinated engagement of actors from the mentioned societal arenas. This comprise of:

1. An analysis of how to address the challenges of staging sustainable transition as societal processes through the use of governance strategies and provide methods to enable stakeholders to make continued adjustments of objectives and means in unavoidably conflict ridden transition processes.
2. A description of how key measures and institutions at different societal levels contribute to transition processes. This involves analyses of: *Labelling and accounting schemes* as measures to stabilize sustainable socio-technical reconfigurations; *guiding visions* as strategies for changing every day practices and coping with conflicting perspectives and expectations; *innovative designs* as devices to align a variety of partly incompatible structures and perspectives; and

planning as a perspective to clarify openings, choices and opportunities in specific circumstances.

3. A characterisation and analysis of 4-6 typical sustainable transition set-ups as complex though recognisable contexts, which are identifiable to actors or constellations of actors in similar situations. The characterisation will focus on patterns of socio-technical interaction, possibilities for action and potential weaknesses that characterize the set-up. This taxonomy of transition processes can serve as an input on how to navigate in relation to future challenges.

Comparisons between approaches to change

While there are possibilities to use visions and scenarios developed in other research projects these visions and scenarios are still the result of research produced within a given disciplinary framework with its specific assumptions and models. Typically are most of the known scenarios for energy systems made under specific assumptions that back-ground some factors and assume these to be either well known or not changing, while other factors are for-grounded and seen as core to the transformation of the energy system.

CEESA acknowledges the fact that a truly sustainable energy system cannot be achieved without taking into consideration the context in which this system operates. Sustainability is not only a question of renewable energy production, but a matter of general environmental concern. It is seen as important not only to optimize individual sub-systems of the overall energy system, focusing for instance on electricity distribution, transport or production. Environmental aspects must be assessed at a system level, and the design and evaluation of energy systems cannot be done properly without comprehensive environmental assessment tools.

The project also analyses e.g. the role of district heating in future renewable energy systems . By use of a detailed energy system analysis of the complete national energy system, the consequences in relation to fuel demand, CO_2 emissions and cost are calculated for various heating options, including district heating as well as individual heat pumps and micro CHPs (Combined Heat and Power). Through energy systems analyses and load-flow analyses, it is determined that if geographically scattered load balancing utilising the regulation ability of hitherto locally controlled plants is introduced while also introducing new dispatch able loads in the form of electric vehicles and heat pumps, electricity transit is enabled to a higher degree than if central load balancing is maintained.

A more detailed analysis of the models used in the project demonstrates that it leaves out how actors engage in conflicting agendas and also sometimes coordinate and achieve larger transitions. Instead the project operates with rather stylized actor profiles and behavioural assumptions.

Rebeca analyses the impacts of introducing biofuels in Denmark, by covering both application and resource sides and addressing a broad range of impacts, i.e. emissions, air quality, health aspects, resource use, land use, and economic and sociological aspects. By describing and analysing different scenarios, the project contributes to the scientific debate on biofuels in Denmark. As an example one paper contains a tentative suggestion for how to take into account the value of changes in price volatility in real world cost-benefit analyses. In the present paper, a method of valuing changes in price volatility based on portfolio theory is applied to some very simple transport-related examples. They indicate that including the value of changes in price volatility often makes very little difference to the results of cost-benefit analyses, but more work has to be done on quantifying, among other things, consumers' risk aversion and the background standard deviation in total consumption before firm conclusions can be drawn.

CEEH has the aim to evaluate and apply integrated models for core impact pathways, including integrated energy systems, emissions, atmospheric chemistry/transport, human exposure, human health models as well as cost models. The models are considered used to optimize the energy production system from a grand economical viewpoint, and will be used to provide qualified guidelines for all sectors of the future energy planning in Denmark. When implementing cost estimates of pollution damage (externalities) from energy production and consumption it is possible to determine the cost effectiveness of air pollution, health effect prevention, mitigation methodologies/technologies, or to compare and optimize the total energy cost options for the society. This include health impact of air Pollution, concentration-response functions derived from the epidemiologic studies in the literature as well as estimating the effect of air pollution in different scenarios for future Danish energy production through demographic and epidemiological modelling. The project will provide valuable data for assessing health impacts but the idea of optimization based only on these impact data seem to have limitations in relation to the dynamic in society concerning regulatory measures and the acceptance of negative impact in comparison with e.g. loss of jobs etc.

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Development of a multi-criteria evaluation framework for alternative light-duty vehicles technologies

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Abstract

This paper describes the process of developing a multi-criteria evaluation framework for alternative fuel/technology options for light-duty vehicles in a mid-term horizon. Such a framework is intended to assist policy makers and governments to take decisions concerning the development of infrastructures or the establishment of incentives to promote alternative fuel vehicles with a mid-term vision.

A comprehensive literature review enabled us to identify the key attributes for comparing different pathways. These attributes were grouped into five main categories: user's acceptance, emissions of pollutants to atmosphere, risk of the technology development, transition cost, and availability of fuel supply. Regarding the alternatives, we can have a large number of combinations for alternative technologies/fuels that can be used in light-duty vehicle fleets. Several fuels such as Ethanol, Methanol, CNG, LPG, Hydrogen, biodiesel, DME, as well as gasoline and diesel, have been investigated. On technologies, beside the current common technologies (the port injection spark ignition engine (PISI), and the direct injection compression ignition engine (DICI)), there are several other technologies such as the direct injection spark ignition engine (DISI), the Hybrid-PISI, the Hybrid-DICI, the Fuel Cell with/without reformer, the Plug-in Hybrid (PHEV), and the Battery Electric Vehicle (BEV).

Multi-Attribute Utility Theory (MAUT) is very popular in the context of multi-criteria decision making because it easily incorporates the decision maker's preferences. The basic goal of MAUT is to replace available information by "utility values" allowing the comparison of alternatives. For the basic MCDA problem of choosing the best alternative, it is useful for a DM to start by eliminating those alternatives that do not seem to be interesting. This procedure is often called "screening". Screening helps by allowing the DM to concentrate on a smaller set that (very likely) contains the best alternative. In this work we have applied a sequential screening process, starting with a Pareto Optimal (PO) approach, followed by a Data Envelopment Analysis (DEA) based screening and Trade-off Weights (TW) procedure.

To illustrate the approach, Portugal was chosen as a case study. Besides, at this preliminary stage of the research, we just considered alternatives with 100% of one specific fuel/technology combination (alternatives with fleets combining different shares of fuels/technologies will be analyzed in the next phase of the research). MAUT was applied to identify the utility values of each alternative for each

group of attributes. Then the sequential screening approach was applied. The final screening set includes DICI-DME, Fuel Cell using Hydrogen, the Fuel Cell with reformer using Methanol, and Hybrid Gasoline. As a conclusion, preliminary results clearly show the potential of the developed approach in setting a framework for supporting better and sounder decision-making on which AFV technologies should be supported.

Introduction

This research aims at developing a multi-criteria evaluation framework for alternative fuel/technology options for light-duty vehicles in a mid-term horizon. Such a framework is intended to assist policy makers and governments to take decisions concerning the development of infrastructures or the establishment of incentives to promote alternative fuel vehicles with a mid-term vision.

In the literatures, several different approaches were introduced to the assessment of alternative fuel/technology vehicles, including Life Cycle Analysis (LCA), Societal Cycle Cost Analysis and Multi-Criteria Analysis.

LCA is a “cradle-to-grave” approach for assessing industrial systems. “Cradle-to-grave” begins with the gathering of raw materials from the earth to create the product and ends at the point when all materials are returned to the earth. LCA enables the estimation of the cumulative environmental impacts resulting from all stages in the product life cycle, often including impacts not considered in more traditional analyses (e.g., raw material extraction, material transportation, ultimate product disposal, etc.) (SAIC, 2006). Ideally, the purpose of LCA is to analyze how the world can change toward the alternative systems. In practice, however, most LCAs does not specify or analyze a policy, but just assume (implicitly) that one simple and narrowly defined set of activities replaces another.

More comprehensively, Societal Lifecycle Cost approach is proposed to compare alternative automotive engine/fuel options from different aspects including the vehicle first cost (assuming large-scale mass production), fuel costs (assuming a fully developed fuel infrastructure), externality costs for oil supply security, and damage costs for emissions of air pollutants and greenhouse gases calculated over the full fuel cycle (Ogden, 2004).

There are however, two critical issues with these two approaches. First, these methods are much depending on the accuracy of the collected data which may result in considerable uncertainty about the conclusions and secondly, we cannot incorporate the attitude of Decision Maker (DM) in the comparison process.

Multi-Criteria Analysis approach can rank alternative fuel-technologies from different aspects including technical, environmental, economical and social. There are several benefits of using Multi Criteria assessment approach:

- Impacts do not need to be monetized; they could be expressed through a variety of measurements.

- Enables to combine experts and non-expert scientific understanding, knowledge and values
- Helps us to illustrate the trade-offs
- The results could be used to aid decision-making when several competing socially and politically criteria are considered

During the literature review phase, it was well concluded that there are numerous factors which could potentially affect the comparison of different pathways. Several factors have been considered in order to characterize the articles, including:

1. Region; the country or region covered by the analysis could affect the results (mainly due to the feedstock characterization and existing infrastructure);
2. Time Frame: the target year of the analysis;
3. Vehicle drivetrain type: Including ICEVs= Internal Combustion-Engine Vehicles, HEVs= Hybrid-Electric Vehicles (vehicle with an electric and ICE drivetrain), BPEVs= Battery-Powered Electric Vehicles, FCEVs= Fuel-Cell Powered Electric Vehicles.
4. Fuels: Fuels carried and used by motor vehicles. FTD=Fischer-Tropsch diesel, CNG= Compressed Natural Gas, LNG=Liquefied Natural Gas, CH2= Compressed Hydrogen, LH2= Liquefied Hydrogen, DME= Dimethyl Ether.
5. Feedstock; the feedstock from which the fuels are made.
6. Vehicle Lifecycle: The lifecycle of materials and vehicles. Apart from vehicle fuel. The lifecycle includes raw material production and transport, manufacture of finished materials, assembly of parts and vehicles, maintenance and repair, and disposal.
7. Green House Gases (GHGs): the pollutants that are included in the analysis of CO2-equivalent emissions, (Equivalency factor approved by the Intergovernmental Panel on Climate Change (IPCC)).
8. Infrastructure: The life cycle of energy and materials used to make and maintain infrastructure, such as roads, buildings, equipment, rail lines, and so on.

Another conclusion from the literature review was that, there are two key different perspectives to tackle the evaluation of alternative fuel-technology vehicles:

- Policy Maker's Perspective (Top-Down)
- User's Perception (Bottom-Up)

In the Top-down approach, the focus is on universal issues such as GHG emissions and security of fuel supply. This approach is adequate when the target is to incorporate the stakeholder or National-wide Decision maker's concern in the decision making process. On the contrary, looking from users' perspective requires the bottom-up approach, in which the focus is more on vehicle characteristics such as first cost, safety and performance. In this research, our aim is to combine these two approaches, by considering the decision maker as an agent of public interest. Therefore, we will include some wide-ranging firms like GHG emissions and fuel supply security, as well as initial cost and fuel cost of alternative fuel/technology vehicles.

Model Setup and Parameter Specification

A comprehensive literature review enabled the identification of attributes for comparing different pathways. These attributes were grouped in five main categories: User's acceptance, Emissions to atmosphere, Risk of technology development, Transition cost, and Availability of fuel supply. Regarding user's "acceptance", two factors were considered: vehicle's lifetime expenses and its' performance. Global GHG emissions and air pollution were gathered in the "emissions to atmosphere" group. In this analysis, the "risk" of the alternative technology not becoming developed, was also assessed. Moreover, the "transition cost", including the investment required for alternative fuel supply systems, was one of the key criteria. "Availability" of fuel supply involves three key factors: resource diversification, variation of energy carriers and share of renewable energies.

Regarding the alternatives, there can be a large number of combinations of alternative fuels and technologies to be used in light-duty vehicle fleets. Several fuels such as Ethanol, Methanol, CNG, LPG, Hydrogen, biodiesel, DME, as well as gasoline and diesel, were investigated. On technologies, beside the current common technologies (the port injection spark ignition engine (PISI), and the direct injection compression ignition engine (DICI)), there are several other technologies such as the direct injection spark ignition engine (DISI), the Hybrid-PISI, the Hybrid-DICI, the Fuel Cell with/without reformer, the Plug-in Hybrid (PHEV), and the Battery Electric Vehicle (BEV). Our approach aims at finding a set of alternatives which are potentially satisfactory, when decision makers (DM) with different attitudes are involved in the process.

Methodology

In order to provide support to decision makers in their search for satisfactory solutions to the multi-criteria decision problem, it is necessary to construct some form of model to represent decision maker preferences and value judgments. Such a preference model contains two primary components:

Preferences in terms of each individual criterion, i.e. models describing the relative importance or desirability of achieving different levels of performance for each identified attribute.

An aggregation model, i.e. a model allowing inter-criteria comparisons (such as trade-offs), in order to combine preferences across criteria.

There were identified three different classes of preference model which can be adopted in considering multiple criteria decision problems including value measurement, satisfying method and outranking approach. Here, a short discussion about utility theory that can be viewed as an extension of value measurement will be presented.

Multi-Attribute Utility Theory (MAUT) is very popular in the context of multi-criteria decision making because it easily incorporates the decision maker's preferences. The basic goal of MAUT is to replace available information by "utility values" allowing the comparison of alternatives. (Daellenbach, 1994). MAUT allows the decision maker to develop reasonable preference criteria, to determine which assumptions are most appropriate, and to assess the resulting utility functions (Lindey, 1985).

For the purpose of developing multi attribute utility theory for each criterion, a quantitative value on a cardinal scale shall need to be determined. The attribute values are, however, not fully determined by the choice of the alternative, but may also be influenced by unknown exogenous ("random") factors. The consequence of each alternative is thus described in terms of a probability distribution for each attribute (Belton and Stewart, 2002). Based on Stewart (1995) conclusions, although the multi attribute utility theory is of value in identifying pathological problems in which additivity may be inappropriate, in practice the use of additive models for decision making under uncertainty is likely to be more than adequate in the vast majority of settings.

After the construction of a basic MCDA problem and the acquisition of preferences from the DM, a global model to aggregate preferences and solve a specified problem (choose, rank or sort) may be constructed. A typical example is the linear additive value function, which can be expressed as:

$$V\left(A^i\right) = \sum_{j \in Q} w_j . v_j(A^i) \qquad \text{(Equation I)}$$

In which, V(Ai) is the aggregated evaluation of alternative Ai, $V\left(A^i\right) = \sum_{j \in Q} w_j . v_j(A^i)$ is the value vector for each sub-attribute j, and the wj is the weight vector of sub-attribute j for aggregation process. In this model, for example, in order to aggregate value of sub-attributes such as vehicle expenses and vehicle's performance into higher level one (Acceptance), the suggested equation will be applied.

For the basic MCDA problem of choosing the best alternative, it is useful for a DM to start by eliminating those alternatives that do not seem to be interesting (B.F. Hobbs & P. Meier, 2000). This procedure is often called "screening". Screening helps by allowing the DM to concentrate on a smaller set that (very likely) contains the best alternative. In this work we have used a multi-stage screening process, starting with a Pareto Optimal (PO) approach, followed by a Data Envelopment Analysis (DEA) based screening and Trade-off Weights (TW) procedure. This seems to be a rather powerful sequential screening technique (Ye Chen, et al., 2008).

Briefly, at Pareto Optimal screening phase, the first step is to identify preference direction (positive or negative) for each criterion; then alternatives will be compared based on their consequence data and then dominated alternatives based on the definition of Domination will be determined; final step is to remove dominated alternatives and retain non-dominated alternatives. Data Envelopment Analysis (DEA) is a technique used to measure the relative efficiency of a number of similar units performing essentially the same task. DEA was first put forward by Charnes *et al*. (1978) who proposed the basic

DEA model, called CCR. DEA based screening starts with the application of a DEA model (usually CCR) in order to identify dominated alternatives; finally, the dominated alternatives will be removed.

Based on Trade-off Weights (TW) Screening, an alternative is potentially optimal (PotOp) if there exists w (weights for each attribute) such that the aggregated evaluation of alternative (V(Ai)) could be the maximum value compared to the other alternatives. The favourable set of alternatives includes all the PotOp alternatives.

Results and Discussion

In this research, a methodology combing MAUT and this multi-stage screening approach is used to identify a set of interesting alternative fuel/technologies. To illustrate the approach, Portugal was chosen as a case study. Besides, at this preliminary stage of the research, there were considered just alternatives that include only one specific fuel/technology combination (alternatives with different shares of fuels/technologies will be analyzed in the next phase of the research). MAUT was applied to identify the utility values of each alternative for each group of attributes (Figure 1).

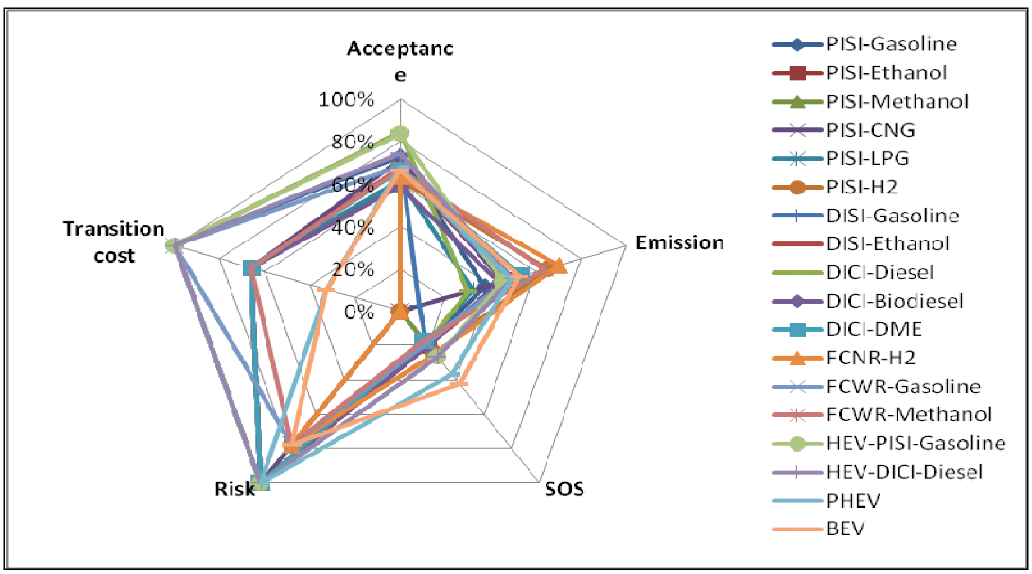

Figure 1: Multi-Attribute evaluation of pure technology alternatives

Then the referred sequential screening approach has been applied. Pareto Optimal screening phase applied in order to remove dominated alternatives (PISI-CNG, PISI-LPG and HEV-DICI-Diesel). Then DEA model enabled us to identify alternatives with lower relative efficiency (PISI-Gasoline, PISI-Ethanol, PISI-Methanol, PISI-H2, DISI-Gasoline, DISI-Ethanol, DICI-Diesel, DICI-Biodiesel, PHEV and BEV). After applying the TW-Based Screening, the final screening set includes the DICI-DME, Fuel Cell using Hydrogen, the Fuel Cell with reformer using Methanol, and Hybrid Gasoline technology. To choose between these options, more information about decision maker's attitude is necessary. As a conclusion, we might say that preliminary results clearly show the potential of the developed approach in setting a framework for supporting better and sounder decision-making.

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A Review of the Climate Simulator of Malmö – a Tool in the Process of Fulfilling Local Mitigation Targets

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Introduction

Since the understanding has deepened of the risks that the climate changes are presenting to the world, the targets set by politicians (at least in some countries and regions of the world) to combat the future effects and mitigate climate change have become more ambitious. Simultaneously, the time to fulfil these ambitious goals shortens and the urge for powerful action is imminent. The City of Malmö is closing up on a great challenge of mitigating emissions. On one hand, the city claim to have a political will to become a leader in local climate work. On the other hand, the city and the region as a whole are growing rapidly. Based on a production perspective Malmö's CO_2 emissions are about to double due to the newly established Öresundsverket (Andrén, 2009), a 440 MW_e and 250 MW_{th} natural gas fired Combined heat and power plant (CHP).

There is also a need for accurate quantifications of the current emissions of greenhouse gases as well as suitable models in order to measure the effects of today's decisions on future emissions, to enable planning, prioritising between measures and follow-up on target achievements. The Climate Simulator provides the City of Malmö with such a tool. The Simulator quantifies current emissions, but it is also looking ahead at different scenarios for the year 2030. A *business as usual*-scenario clearly illuminates the gap between mitigation actions planned today and what needs to be done for the City of Malmö to become a so called climate neutral city. An alternative scenario allows the user to experiment with different measures and conditions relatively freely and create his/her own normative scenario.

The questions that I seek to answer are foremost if the Climate Simulator is a useful tool in the process of fulfilling the climate goals of the City of Malmö. I have defined "useful" by a model which gives an appropriate description of the reality and also is applicable by Malmö in their climate work. Furthermore, I will discuss in which directions the Climate Simulator could be developed to better serve these purposes.

The Climate Work and Climate Impact of Malmö

Malmö is the third largest city in Sweden with the ambition to be in a front position in local climate work. Their Environmental Program for 2009-2020 states that in the year of 2020 the municipal organisation will be climate neutral and by 2030 the goal is for the entire Malmö to be 100 % supported by renewable energy sources (City of Malmö, 2009A). During 2009, the City of Malmö further showed its commitment to the climate issue through entering the Covenant of Mayors, the aim of which is that the signatory cities go beyond the CO_2 reduction targets set by the EU energy policy (European Commission, 2010). Malmö has adopted the same mitigation target as the national goal for Sweden: a 40 % reduction of non EU-ETS greenhouse gas emissions until the year 2020 as compared to the year 1990. A related objective in the Environmental Program of Malmö is to reduce the per capita energy use by 20 % to 2020, compared to the average annual usage of 2001-2005, and with another 20 % to 2030.

These official documents clearly state the political commitment to reduce the negative climate impact. Nevertheless, the changes that need to be implemented throughout the society are massive and even more so by the establishment of Öresundsverket, owned by the energy company E.ON. Looking back at the past CO_2-emissions of Malmö, there was a large decrease of emissions during 1980-1990 (figure 1), which can be explained by the deindustrialization of Malmö during that period and improvements of the district heating system (City of Malmö, 2009B). Between 1990 and 2008 the CO_2-emissions have been reduced with another 7 % (City of Malmö, 2009B), but this negative trend is likely to change when the emissions of Öresundsverket are taken into account. The emissions of CO_2 in Malmö were approximately 1.1 Mton in 2008 (Bruun Månsson *et al.*, 2009) and when Öresundsverket is fully operational they can be expected to double (Åberg, *et al.*, 2009).

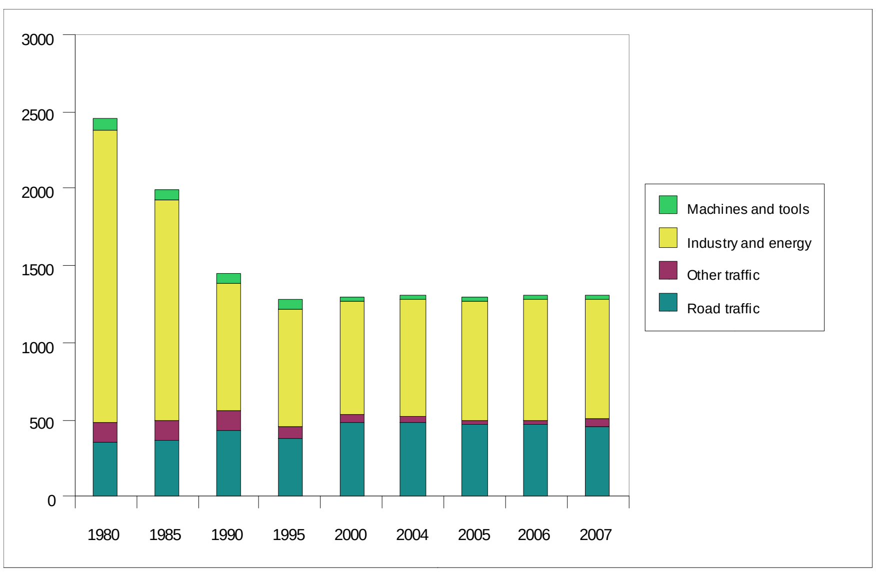

Figure 1: The emissions of CO_2 in kton from different sectors in Malmö during 1980-2007. (Data from the Environmental Department of Malmö).

A set of good examples of sustainable development can be found in Malmö, such as the offshore wind farm Lillgrund in the Öresund strait outside of Malmö, with a production capacity of 0.33 TWh elecricity (Andrén, 2009). However, despite these good examples the main components of the energy system are still highly dependent on non-renewable energy. According to Andrén (2009) non-renewable energy, including nuclear power, supplied 2/3 of the City's energy supply in 2006. The local production of district heating and electricity is mainly fuelled by natural gas and waste.

Description of the Climate Simulator

In its climate work the City of Malmö has experienced a need for heightened knowledge of necessary measures in order to reach their climate goals by the year of 2030. Therefore, the consulting firm Goodpoint AB recently delivered the Climate Simulator to the Environmental Department of Malmö. The excel file called the Climate Simulator can be used for modelling of possible future emission and energy scenarios in Malmö (Bruun Månsson *et al.*, 2009).

The creators have combined data of the energy use with values of emission intensity to calculate the climate impact of heating, electricity, transports, shipping and machines within Malmö. Note that only CO_2-emissions are included in the model. The situation of 2008 has been chosen as a reference scenario, with CO_2-emissions of 1.1 Mton.

The Climate Simulator also provides two alternative future scenarios of 2030: one descriptive *Business-as-Usual*-scenario and one more normative *Backcasting*-scenario. In the *Business-as-Usual*-scenario the different sectors in the energy system are considered to continue developing as they do today, except that the measures already planned by the municipality are also taken into account. In the *Backcasting*-scenario on the other hand, the user can experiment freely with different parameters needed to reach the climate targets of Malmö. The pedagogic effect of these two scenarios beside each other is evident – even though today's planned measures are implemented, it will be far from enough to reach climate neutrality. In both scenarios the energy use of the different sectors of the energy system has been changed proportionally with the estimated population growth of the area.

The model is divided into two main parts and in both of them the values can be changed and experimented with. The first part, the Basic Settings, consists of the external factors affecting the final emissions, emission intensity factors and other more detailed figures. The second part, the Climate Simulator, describes different measures that can be done by Malmö and it also shows to which degree the emissions would decrease when a measure is taken. This part of the model is supposed to be more easily accessible and it describes the key factors affecting the total emissions that are controlled by the City of Malmö.

The chosen perspective on emissions is mainly a geographical production perspective. This is certainly true for transportation, where only emissions from vehicles within the borders of Malmö are included. However, the perspective is much more vague regarding heating and electricity. In the model the total energy needed for heating and electricity is calculated from the energy demand in the buildings of Malmö, which means a consumption perspective, while the sources of energy are assumed to be equal to local production. Since the establishment of Öresundsverket, Malmö will become a large net exporter of electricity and therefore could be considered as self sufficient in power. The shares of different energy sources can be experimented with in the model, but the production with the highest carbon intensity is set to be the source filling up the remaining share, both for district heating and electricity use.

In the Climate Simulator the goal of Malmö is to become climate neutral, in the sense that it needs to decrease its CO_2-emissions as much as possible and compensate for the rest. Compensation to a certain extent will always be needed, since it is not possible for Malmö to decrease its emissions to zero only by local measures – there will always be some emissions from the renewable energy supply, which are included in the model. The City of Malmö is not allowed to invest municipal tax money in climate mitigation projects abroad, e.g. CDM projects to receive CER:s. Instead, the model offers compensation for the Malmö emissions from the supposed annual off-setting by the Swedish government, estimated to be 6.7 Mton of CO_2-equivalents according to the goal for 2020 in the government climate bill (Bill 2008/09:162). Malmö's share of the total off-setting is assumed to be the same as its share of the population which is about 3 percent.

Conclusions

Let us now return to the questions formulated in the beginning. *Is the model an appropriate description of reality?* First of all, some emissions are excluded from the model, such as emissions outside of Malmö induced by consumption of the Malmö inhabitants, other greenhouse gases than CO_2, which are said to represent only 5 % of the total emissions (according to Lars Nerpin, Environmental Department), emissions from airplanes and also emissions from the production of exported electricity at Öresundsverket. All of these emissions could also be said to be the responsibility of Malmö. To include them all in one model would probably be difficult, still it is important to keep track of them and keep them in mind. Those engaged at a political level, or at the Environmental Department, also need to clarify which emissions should be included in the targets. My conclusion is that the model provides one of several possible ways of looking at reality of today. The way of looking at future energy sources can be found to be a bit unimaginative, since only technologies available today are taken into account, and with a kind of overconfidence in the potential of biogas, but it can still give us valuable information about which direction Malmö is heading.

Is the model applicable by Malmö in their climate work? The mix of production and consumption perspective on the emissions is a bit confusing for the user of the Climate Simulator, but it can be motivated by the focus on measures that the municipality can control. The model must show an effect of the measures taken. If it were a purely production perspective, the establishment of new windpower capacity would only increase the total emissions, whereas in the Climate Simulator the windpower substitutes the fossil power from Öresundsverket. Therefore, the increased windpower capacity it is the one single measure that brings on the biggest emission reduction in the model, according to my sensitivity analysis.

Moreover, the Climate Simulator needs to be used with background knowledge and an active critical perspective to always question the plausibility of the modelled scenarios. When used in this way it can be a useful tool to show the gap between measures planned and measures needed. My main critique of the Climate Simulator is, however, that the target level has been defined differently compared to the official documents of the City of Malmö. A clearly defined target level is crucial in backcasting and according to the Environmental Program and the Energy Strategy of Malmö the goal is to completely support the energy system with renewable energy and not to minimise emissions and compensate for the rest. The weakness of the Climate Simulator in this context is that it works well to analyse only emissions, but it does not give more information about the potential of biofuels or how big the energy use per capita is.

Last of all, I will discuss the possible development of the model. Certain parts, especially regarding motorized transportation, could be more detailed, including possibilities to experiment with shares of different drivelines and fuel inputs, in order to match the already more detailed data of heating and electricity use. Here the emissions from cars should be prioritized, since it is the second largest emitting sector in the model. To further detail the model, the other greenhouse gases could also be included.

Another way of developing the Climate Simulator would be to add more graphic, thus turning it into a pedagogical instrument, not only for internal use. The model could also be adjusted and used in other municipalities, which would improve the comparability between their quantifications of emissions. However, if the model were to be used in other municipalities, be careful not to exclude any important emissions or for example double count renewable energy. A more widespread use could also imply the need for the municipalities to have more similar goals. If the Climate Simulator could be developed to include more dimensions than just the reduction of emissions, such as decreased energy use, that would also be interesting, since it would show how these dimensions are related to each other.

The work of quantifying emissions on a local scale, especially regarding electricity, since the difference between local production and local consumption can often be huge, is not without problems. Neither is the creation of future scenarios an easy task. Nevertheless, this work is essential in combating climate change and to enable long term planning. However, it should be conducted along with a critical review of these models, while they are still in a stage of development and their limitations cannot be disregarded.

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The Application of Sustainability Criteria in defining conditions for an environmentally adapted development of biofuels in Denmark

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Introduction

The market of biofuels for transport has been set in motion around the globe and is experiencing a rapid development spurred by targets and policies set in place in OECD countries. The motivation has being to harness biofuels potential contribution to greenhouse mitigation, energy supply security, support for agricultural industries and reduction of oil imports. The Euroean Union has set a target for all transport fuels in Europe to be 10% from renewable energy sources by 2020. In Denmark the goal has been set to 0,75% biofuels --in gasoline and diesel- by 2010; 3,3% in 2011 and 5,75% by 2012 (Teknologirådet, 2009).

The rapid evolution of global biofuel for transport demand creates a pressure to design, implement and verify sustainable biofuel production practices. The present study explores elements of sustainability debate as they apply to the Danish domestic biofuel production within the scope of the REBECa project. Specifically the study reviews and discusses the implications of the mandatory provisions from the European Union directive that sets sustainability criteria for biofuel production, it discusses those requirements and considers those already established in Danish legislation regarding agricultural, land use and nature protection practices, to provide an overview of what may be missing areas where the need for science based knowledge is more urgent to provide a framework to evaluate and accounting for the sustainable biofuel production practices at the national level.

The REBECa project was set to assess the general impact of the introduction of biofuels in the Danish road transport sector through the development of diverse scenarios from 2007-2030. The project has investigated the environmental, health impacts and welfare economic consequences of using energy carriers based on petrol and diesel mixed with their bio-based counterparts in the domestic market. The study set up an integrated system forecasting road transport in Denmark, fuel demand and demand for raw materials until 2030 based on the Danish policy targets for biofuel use, and developed land use scenarios for biomass production followed by a detailed well-to-wheel analysis to determine environmental impacts air and climate emissions attributable to the expected level of future biofuel energy use in each scenario. While the detailed well-to-wheel based environmental impact analysis in REBECa is unique, contemplation of the EU mandated sustainability analysis and of a number of interconnected sustainability issues at the global level began to provide information that can help to answer questions such as: which biofuels will be good or bad to Denmark? What sustainability conditions on biofuels production should Denmark set up for domestic and imported biofuels?

Sustainability Criteria for Biofuels EU Directive

The EU Directive 2009/30/EC establishes the principle that biofuel production should be sustainable (Art. 8). In practice the directive formulates the expected goals of biofuels production in Europe, for they should provide: greenhouse gas reduction; energy security and self-sufficiency; job creation and economic growth in concrete quantifiable terms. Reporting on the result of its implementation the EU Commision shows for example that net GHG savings have been achieved in the EU from biofuels placed on the market and consumed in 2006 and 2007 amounting to 9.7 and 14.0Mt CO2-eq respectively (EU Comission Communication 2009). Similarly, regarding the goal of security of supply: ”in 2007 the use of biofuels in the EU replaced 1593 million litres of gasoline and 7730 million litres of diesel”. Finally, regarding economic growth the report shows that ”agricultural activity related to the renewable energy sector generates gross value added of well over €9bn per year”. All of these gains offer a clear incentive to supporting the further expansion of biofuel markets for transport in Europe.

The directive introduces mechanisms to monitor and reduce GHG emissions from transport; adapts methodological principles and values necessary for assessing sustainability criteria; establishes criteria and geographic ranges for what are highly biodiversity grassland areas; and adapts the methodology for calculation of lifecycle greenhouse gas emissions of fuels used in transport. Regarding who should perform the monitoring and keep track of required indicators the directive states that its mandatory requirements apply to single consignments of domestic or imported biofuels regarding: feedstocks, final products and production methods. Similarly, the reporting needs to be made directly by the companies involved while the enforcement of criteria is the responsibility of each Member States (including adequate standards of independent auditing). At the EU level members need to report to the Commission every two years through submission of a written report.

The mandatory criteria are presented in the summary table 1 below. A number of issues such as the indirect land use (iLUC) effect (art.14): use of restored degraded land and increase land productivity (art22) are only promised to be dealt with through construction of a ’factor’ to plug in the GHG calculations which will be announce in a Communication on the Practical implementation during 2010. Other definitions that will need further elaboration to become meaningful sustainability criteria are: the definition of degraded land, biodiversity, grassland -updating of default values -guide on carbon stocks -reviewing impact of iLUC

Regarding implementation of this directive the Commission recognizes ’voluntary schemes’ and international agreements and member states should accept this evidence. The enforcement of the criteria however is responsibility of each Member State. The criteria apply to both domestic production and imported biofuels. Members should establish national or regional averages for emissions from cultivation, including from fertiliser use. This area is one of the areas in which REBECa has already contributed to the development of a national sustainability framework for biofuels. The work developed in the REBECa study has also importance as the directive establishes that members are encourage to draw up their own tables and make them public. Currently the status of biofuels domestic production in Denmark is still minimal, however a number of specific tasks will need to be advanced as all member states are required to submit a report every two years to the Commission giving a fair status of: land use, soil, water, air protection, impact on biodiversity, social sustainability, food and others.

The Directive does not penalize or is capable of preventing unsustainable biofuel production. The verification procedure defined in the Directive is still in evolution. In short the verification of compliance with sustainability criteria is defined through a mass balance system: (art 7c) –which allows consignment of raw material with different sustainability characteristic to be mixed. The information refers to the mix and what finally matters is that the sum of all consignments can be described as having the same sustainability characteristics. Further elaborations on how this mass balance system can operate have been advanced as it has been proven in Germany where an International Sustainability and Carbon Certification System has been put into place, heavily supported by independent auditors and so far implemented with success in pilot audits in Argentina, Brazil, Malaysia and Europe (World Bioenergy Conference report, 2009). Another example of full investment in the verification of the sustainability of biofuel production can be found in the United Kingdom where a new agency was created the Renewable Fuel Agency as an independent sustainable fuel regulator to administer the "Renewable Transport Fuel Obligation Order" (RTFO) since 2007 (Renewable Fuel Agency, 2008).

To reward good behavior of economic actors, the EU Directive (art16), specifies that those biofuels that meet the sustainability criteria could command a premium price. To exemplify the difficulty of verification the UK Renewable Fuel Agency has the power to impose Civil Penalties of up to 50,000 pounds for either evasion or submission of inaccurate data but establishes no legal penalty for companies failing to comply with sustainability standards. However, the agency openly seeks to report in its monthly magazine and website the results of its company auditing establishing a principle of public name/shame which may encourage further compliance of actors. Still according to a 2008/9 annual report from the agency 73% of reporting did not meet environmental requirements of RTFO that year.

Global Trade Verification of Sustainability of Imported Biofuels

Uncertainty and gridlock characterizes the global biofuel markets. The status continues to be one in which there are no clear routines for lending banks, no established traditions for trading between fairly new institutions with new mandates; there is high degree of susceptibility in trading due to disinformation, and no links between good environmental performance and financing (World Bioenergy Conference report, 2009).

In the international scene of biofuel producers at present only Brazil is posed to pass verification and to have the capability to withstand international certification. The same cannot be claimed from all nations in Asia where a number of issues regarding the indirect land use iLUC, competition with food production, and biodiversity need to be clearly justified. In the United States, the passage of a Farm bill in 2008 giving a substantial subsidy for the production of cellulosic ethanol raises also the stakes of unproven issues with iLUC, food and biodiversity issues. Finally, the African continent at large continues to lack one or more of what have been called the 4Is: Infrastructure, Investment, Institutions, Implementation capacity which limit the possibilities of assuring the verification of sustainability biofuel production there. (Renewable Fuel Agency, 2008)

This short review of the international verification possibilities of the sustainability of biofuel productions shows the extent to which the present schemes spurred with the EU Directive have a "Eurocentric" character that may be short of delivering the intended sustainability of biofuels at a international or global scale. As the major producers of biofuel may be all in developing countries there is a legitimate concern that their stakeholders concerns do not figure in the schemes with the same prominence as developed countries concerns over GHG reductions or energy security and access in the formulation of sustainability criteria. Developing country concerns include among others: farmers rights to the land, land use changes (in terms of tenure not only carbon), access to water supply, soil erosion, inclusion of communities of small farmers and especially women as part of the economic model (poverty vulnerability), low cost for land and labor etc. These are all tangible and quantifiable effects that while invisible today may turn out to be like a can of worms if not addressed by the current sustainability evaluations. No company in Europe will be interested in being accused of increasing hunger, poverty or both. A set of national interpretations are needed and new knowledge in the formulation of science based policy is needed. These set of issues also highlight the importance of defining a framework to help in the sustainability assessment of imported biofuels

Good or Bad Biofuels can this question be answered?

According to the discussion above and the relevance of projects like REBECa, an answer to this question can be formulated from the point of view of what are the relative GHG emission savings of different feedstock, production and utilization practices. However, considering biofuel sustainability in general and compliance with specific criteria the challenge is greater. Measuring indirect effects such as the iLUC effect and others may prove to be the most difficult in terms of demonstrating sustainability production. Given the interconnected nature of these issues, the solution will require that tradeoffs are made at the national level of decision making. Therefore what is to be part of a tradeoff will need to be explicitly considered in a framework that allows comparison and evaluation of different outcomes.

Additonal conditions to help define what can be term "good" biofuel production (Bõrjesson, et all, 2009) includes: Biofuel plants should run on renewable energy, not fossil fuels; areas of cultivation are avoided on "carbon rich" soils (i.e: peat soil); possible by-products should be utilized effectively (optimize indirect energy and climate benefits); effective fertilization strategies (N2 0 cleaning equipment); biofuels should to a large extend are best when produced in combination with electricity and heat. Table 1 below summarizes elements that need to be accounted for in the task of formulating a framework to evaluate sustainability of biofuel production in Denmark.

Conclusions

Biofuel for transport development is taking place at a faster rate than countries like Denmark would seem to be prepared to react institutionally. Assuring the sustainability of biofuel production at the national level will depend on scale, pace of development as much as it will on the verification of sustainability criteria for domestic and imported biofuels alike. The Danish authorities will need to take some concrete steps in the short term to comply with EU Directive targets and compliance reporting. The REBECa project has already advanced a number of concrete values that can help pave the way to creating a framework for evaluation. The scale and pace of development can be politically decided as the cases of Sweden, Germany and United Kingdom are demonstrating. Systemic strategic shifts toward implementation of a variety of renewable fuels for transport may have more promising results to ensure sustainability than single piece meal biofuel target implementation.

	General Goals/ Areas where evaluating indicators for a framework exists
EU Sustainability Criteria	GHG saving of at least 35% -50% from 2017 - 60% for new installations from 2018 - default values and calculation method for actual values included No raw material from converted land with: -- high biodiversity value --high carbon stock Chain of custody must respect mass balance methodology Biannual reporting on: Land use ; Soil, water and air protection measures; Impacts on biodiversity; Social issues; Food security, wider development issues; Commodity price changes
Danish Relevant Legislation	-Environment Objectives Act (2003 with later amendments), implementing the Water Framework Directive and the Habitats Directive -The Nature Protection Act (2009) -The Environment Protection Act -The emission ceilings regulation (2003, implementing the NEC directive)
Danish National Green Growth	-Improving water quality -Reduction of damage caused by pesticides,

Strategy	-Reduced ammoniac use -Reduced greenhouse gases -Better nature conservation and biodiversity -Accessible nature improved -Improved oversight of the Danish environment (green Denmark mapping)
REBECa environmental impacts	-Emissions from driving (NOx, PM, VOC, CH, CO) -air quality (PM2.5, PM10, geographically divided) -W-T-W emissions (CO2-CH4 N2O, CO2eq, NOx, SO2, CO, NH3, PM) -land use (need vs available) -emissions from the production of raw materials (N, P, pesticidtryk) -landscape (visual effects of perennial crops)

Table 1: Variables and areas to be considered in a framework to evaluate sustainability of biofuels.

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Optimal Investment Paths for the Danish Energy System in the CEEH Modelling System

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World Energy Outlook 2009 (IEA 2009) projects world primary energy demand to grow on average 1.5% annually in the period 2007-2030. Depending on which energy sources the demand is covered from and the accompanying technologies the resulting welfare of society may differ significantly. Among the reasons behind this is various negative indirect effects on environment and health (external costs) which energy systems cause. They are primarily due to burning of fuels leading to emissions of substances like particles, CO2, NOx, SO2 etc. Global Climate Change is another a good example of an externality illustrating the significant impact energy systems may have.

Fossil fuel related air pollution influences both the natural environment and human health. The particle pollution from cars and trucks alone is considered to cause more deaths than traffic accidents. This has lead to the establishment of a Danish Centre for Energy, Environment and Health (CEEH) which is supported by the Program Commission for Energy and Environment under the Danish Council for Strategic Research. The objective of CEEH is to establish an interdisciplinary based system to support optimal future planning of energy production and usage with respect to costs related to the natural environment and human health. To ensure the needed interdisciplinary approach the centre includes researchers from meteorology, air pollution, environment, energy, physiology/health and economy. The main outcome of the centre is an integrated regional model system including components for air pollution chemistry and dispersion down to urban and sub-urban scales, and model components of the impacts on public health and the external environment.

A key element of the CEEH is to expand, evaluate and apply integrated models for all impact pathways, including integrated energy systems, emissions, atmospheric chemistry/transport, human exposure, human health models as well as cost models. This chain of models is necessary to optimise investments in the energy system from a grand economical viewpoint, and will be used to provide qualified guidelines for future energy planning in Denmark within all sectors. When implementing cost estimates of pollution damage (externalities) from energy production and consumption it is possible to determine the cost effectiveness of air pollution, health effect prevention, mitigation methodologies /technologies, or to compare and optimise the combination of energy options for the society.

Estimates of the cost of emission of SO2, NOx, PM2.5 and CO are calculated in the CEEH project and documented in Brandt et al. 2009. New updated health cost estimates will also be presented at the conference by other project participants. Additionally, different levels of the CO2 price (the global cost by emitting GHG) are assumed. The optimisation of the energy system is carried out by means of Balmorel[8] the energy system model used in the CEEH modelling framework. The model includes the Danish heat and power sector, heating of buildings, industrial processes and transport. The surrounding countries are modelled less detailed but includes district heat and power production.

This paper presents some future explorative scenarios (Nielsen and Karlsson 2006) for the Danish energy system until 2050. The scenarios will explore some socio-economic optimal investment paths for the Danish energy system (including health externalities) given a certain global framework. The global framework will be based on the IPCC RCP 6.0 inventory (Representative Concentration Pathways, pre-scenarios to develop new scenarios for IPCC AR5). The RCP's are inputs to the climate modeling for IPCC AR5. Four scenarios are provided by four modeling groups: MESSAGE (IIASA), AIM (NIES), GCAM (PNNL), IMAGE (PBL), AIM/RCP group.

Name	Radiative Forcing	Concentration	Pathways Shape
RCP8.5	8.5W/m^2 (in 2100)	<= ~1370 CO_2-eq	Rising
RCP6.0	~6.0W/m^2 (stabilization after 2100)	~850 CO_2-eq	Stabilization without overshoot
RCP4.5	~4.5W/m^2 (stabilization after 2100)	~650 CO_2-eq	Stabilization without overshoot
RCP3-PD	< 3W/m^2 (peak and decline) ⇒ 2.6W/m^2	< ~490 CO_2-eq	Peak & decline

(IPCC 2009)

These global scenarios is build on assumptions about some main drivers for the global economy, which is used in the global integrated assessment energy models to calculate future energy demands, production and emissions. The different model groups have then made projections in their respective models.

As global reference for the CEEH modeling system the RCP4.5, referring to a 4.5 W/m2 increase in radioactive forcing, is used. This means that the CEEH air transport models will use emission data for the Northern hemisphere as input for the period 2000 until 2050 as background emissions before running the more detailed energy models of Denmark and surrounding countries. From these global scenarios also future fuel prices can be derived and transferred to Balmorel.

[8] See www.balmorel.com for full model documentation

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Bioenergy yield from cultivated land in Denmark – competition between food, bioenergy and fossil fuels under physical and environmental constraints

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Abstract

Globally, bioenergy is emphasized as an important contributor to reach strategic goals of energy security. The commodity markets for energy, bioenergy and food are interdependent and interacting through the energy dependency of agriculture, an increasing demand for both food and energy, and the option to replace fossil energy resources with bioenergy resources.

A model for supply of biomass for bioenergy in Denmark was developed using linear programming. The model includes biomass supply from annual crops on arable land, short rotation forestry (willow) and plantation forestry, and minimizes production costs of an energy mix consisting of bioenergy and fossil diesel oil. Here, we analyze the possibilities of substituting domestic bioenergy for fossil energy under the constraint of a given food supply and environmental constraints on land use.

Crop area distributions of a total area of 3200 kha were simulated in two sets of scenarios, each examining a range of fossil oil prices. Both scenarios were based on cost and production data of the year 2005. Scenario (a) required a total food&feed energy yield similar to that produced in the year 2005; scenario (b) addressed high prioritization of dedicated bioenergy crops. This was secured by relaxing the food&feed supply to 50% of the 2005 production level. Further, a maximum limit of 25% cultivation area with willow in short rotation was set, and the area reserved for permanent grassland was set to 275 kha (+100 kha compared to 2005). The trade-based animal husbandry sector was excluded from the analysis and the forest area was fixed to 600 kha.

The crop area distributions were affected by fossil oil prices varying from oil index 25 to 200. Oil index 100~9.4 € GJ^{-1} corresponded with a crude oil price of 55$ per barrel in 2005. The woody biofuels, especially high-yielding willow in short rotation, were competitive with fossil oil from around oil index 40 and occupied the maximum allowed area in all crop area distributions, except the model optimized with

oil index 25. In contrast, no land was allocated for bioenergy from oil seed rape and sugar beet cultivation at oil prices below oil index 170.

The analysis shows that the potential for replacing fossil energy with bioenergy is lower than 19% of the primary energy demand if the bioenergy is based on domestic biomass production. A further increase in the use of bioenergy relies on imports from world market supplies.

Introduction

Bioenergy is foreseen to be an important part of future energy supply. Following the oil crises in the 70'ies bioenergy products from forestry, and later, agricultural residues have become part of the energy supply. The bioenergy consumption in Denmark reached 100 PJ in 2005 (Danish Energy Agency 2006). Future perspectives of diminishing fossil oil reserves and thus volatile and increasing oil prices have made bioenergy an attractive alternative. The energy efficiency of plant based bioenergy depends on land productivity, cultivation, and conversion methods.

The question now remains in what quantity and at what cost bioenergy from different crop types can be supplied from land cultivation, and how the fossil oilprice interacts with biofuel costs. The question may be analyzed as an optimization problem.

Materials & methods

The analysis is based on a comparative static cost minimization model for bioenergy feedstocks grown in Denmark using currently grown crop classes and yield levels, Table 1. A detailed description of the model and its parameters may be found in Callesen *et al.* (2010). The model uses linear programming for providing solutions to an objective function that minimizes the cost of a fuel mix of bioenergy and fossil oil, represented by diesel oil, by changing the crop area distribution.

Feedstock type	Crop representative	Conversion method and efficiency
Woody lignocellulosic	Norway spruce, yield level PK8 and PK12 in 60 yr rotation Willow in short rotation forest (22 yr) on sandy and loamy soils.	Heat and combined heat and power (69-81%)
Grassy lignocellulosic	Grass-clover ley with 30-50% clover	Biogas (54%)
Oil crops	Oil seed rape on sandy (JB1-3) and loamy soils (JB5-6)	Rape Methyl Ester (70%)
Starch crops	Winter wheat on sandy (JB1-3) and	1st generation bioethanol (57%), straw used in

	loamy soils (JB5-6)	combustion (90%)
Sugar crops	Sugar beet on loamy soils	1st and 2nd g. bioethanol (54%), tops used for biogas (54%)

Table 1: Feedstock types, crop representatives, conversion methods and efficiencies used in the model.

Data on crop yields, input factors and input prices from the year 2005 were used. A key issue in the model is the changes in real fossil fuel prices and its influence on other costs of inputs used in the cultivation. The cost price increase as a share of the real oil price increase was based on an evaluation of the direct and indirect energy used in the production of these input factors: seeds 25%, fertilizers 50% (nitrogen and potassium) or 75% (phosphorus), lime 50%, machines 25%, fuels and lubricants 100%, pesticides 25% .

Constraints on crop area use were delineated based on land data, limitations due to crop rotation requirements, protection of forest area (600 kha evenly distributed on average and low productive soil), permanent grassland (175 kha) and other constraints set by biological requirements of the crops.

In this application of the model we analyzed two sets of scenarios each with fossil oil prices ranging from index 25 to index 200 in intervals of 25 (<oil index 100) or 10 (>oil index100):

(a) A set of scenarios based on the food&feed production in 2005 (=100%). Cost levels were as experienced in the year 2005. The food, feed and timber demand was set to 167 PJ starch crop yields, 6 PJ oil crop yields, 11 PJ sugar crop yields, 38 PJ grass for feed based on 2005 crop yields. The reservation of timber for wood products was 5 PJ corresponding to about 25% of wood fellings. The area with willow was 0.2% of the available land area.

(b) A set of scenarios with only 50% food and feed reservation as a variant of the scenario (a). Constraints on area use were due to prioritization of bioenergy and the environment. The short rotation willow was restricted to a maximum of 25% of the land area.

Results

The feedstock cost reflected the combination of yield level (soil quality) and crop type and the cultivation intensity applied. Figure 1 indicates the cultivation cost in € GJ^{-1} of the various crop types after conversion to biofuel as a function of the fossil oil price. The woody lignocellulosic feedstocks had the lowest cultivation cost per GJ produced, whereas grass and low yielding wheat and oil seed rape on sandy soils showed the highest cost at oil index 100 (Figure 1). The fossil oil price intersected the biofuel costs in the range from about oil index 40 to 200.

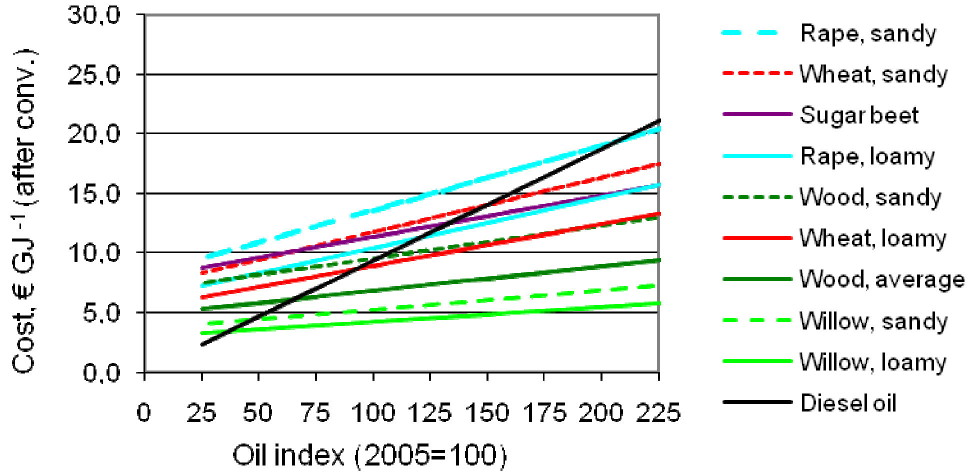

Figure 1: Comparison of costs for producing feedstocks for different biofuels at increasing oil prices (black line, 1:1~ fossil oil index: fossil oil price).

In the (a) scenarios the cost minimized crop area distribution reflected the food constraints laid down in the model and the very limited occurrence of willow plantations. Oilseed rape was grown on a very limited area, and sugar beet was only relevant for bioenergy beyond oil index 160 (Figure 2). In the oil index range from 75 to 150 the biofuel costs for wheat per GJ final energy were quite close to the fossil oil index. Different crop area distributions as solutions to the cost minimization may therefore result in quite similar values of objective function. The suggested optimized wheat area is a range, since fallow land, wheat and fossil oil compete in this price range.

In the (b) scenarios (Figure 2), no crop area was allocated to willow at oil index 25. With increasing fossil oil index the effect of the low yield level was evident since willow on sandy soil was only present at oil index 50. In the remaining price range the maximum willow area was grown on the high yielding loamy soils. Oilseed rape on sandy soil and sugar beet exceeding the mandatory area reserved for food did occur, but only at a relatively high oil price beyond oil index 160 with a concurrent reduction in wheat on loamy soils.

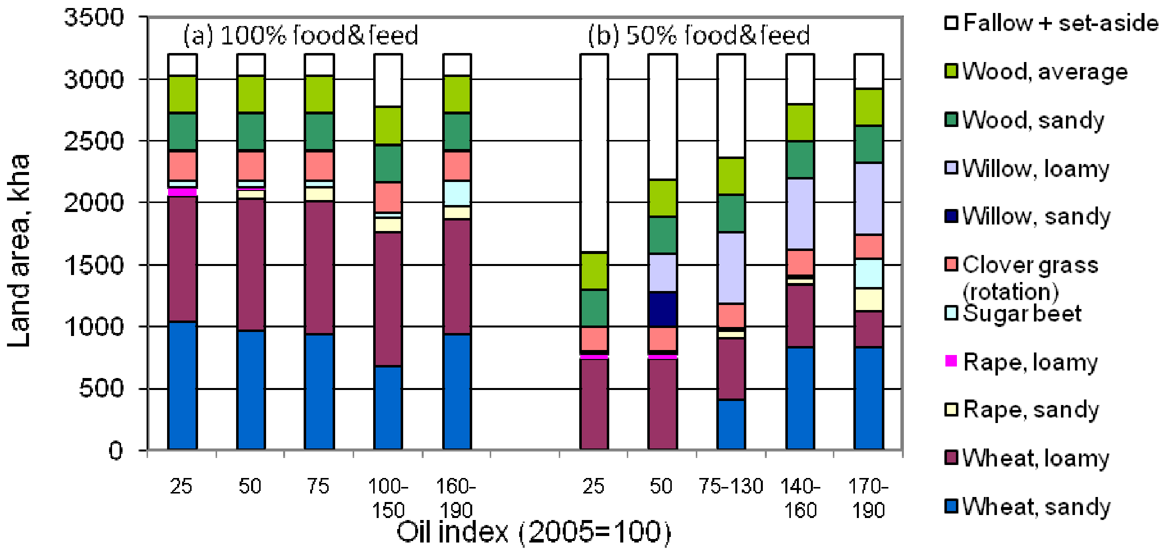

Figure 2: Area distribution of the (a) scenarios with 100% food&feed and the (b) scenarios with 50% food&feed for oil index 25 to oil index 190. Willow is allowed to occupy 25% of the crop area in the (b) scenarios.

The total bioenergy supply across fossil oil prices in the (a) scenarios in Figure 2 ranged between 40 PJ and 60 PJ per year, and the (b) scenarios between 30 PJ and 160 PJ per year. The amount of N fertilizer used in the resulting area distribution of oil index 100 in (a) and (b) (Figure 2) were 375 kt N yr^{-1} and 229 kt N yr^{-1} representing a 146 kt difference in N fertilizer use.

Discussion

If the reservation of land for food supply is decreased, much more land would be set-aside or planted with forest in short or long rotation. The environmental benefits for the environment by reducing nitrogen loads (Erisman *et al.,* 2008) through cultivation of perennial woody crops or setting land aside are obvious. Guesses of the potential available area for willow are 100 kha – far lower than the 581 kha that occur in the model result (Figure 2). There is no knowledge base for large-scale willow cultivation in Denmark indicating if the actual yields and costs can be sustained over time, and if it is accepted by the public. Willow plantations may be a way of increasing the forest area in the long term.The energy sector and the agricultural sectors are regulated, taxed and subsidized in numerous ways. The analysis indicates that volatile oil prices are contributing to the uncertainty of price developments for both food&feed and bioenergy markets. The market for solid biofuels such as wood chips and energy grain is well established and flexible. Switching between different biofuels and co-firing with fossil fuel in both small and large heat and heat and power plants is possible. In comparison with an annual total primary energy use of 800-850 PJ the bioenergy supply would range from 4% to 19%. In addition, there is an unused bioenergy potential, especially from waste in the animal husbandry sector.

In the competition for biomass feedstock, solid woody biofuels had an advantage over liquid biofuels. This may call for market incentives for liquid biofuels (OECD-FAO, 2009). Mandatory blending of biofuel in petrol in the EU27 has been decided by the European Union. Targets are 5,75% in 2010 and 10% in 2020 (European Commission, 2003). Our analysis shows that liquid biofuels will rely on policy mandates, e.g. biofuel blending requirements in petrol.

Long term increases in oil prices speak in favour of biofuels, since most feedstocks can be produced at a competitive cost above the fossil oil index 190. Fossil oil dependent price increases deviating from our assumptions may change the picture. The competitive strength of liquid biofuels will rely on world market supplies of low cost sugar cane or corn based ethanol. The domestic crop land is available provided that feed supplies at a large scale can shift from primary feed to processed by-products such as dried distillers grains with solubles (DDGS).

Conclusion

Domestic bioenergy feedstock production is very limited in comparison with the energy consumption. The possibilities of a substantial increase e.g. by cultivation of willow, even up to 25% of the available crop area, will not increase the bioenergy supply substantially, but the landscape would change dramatically. Biomass imports are needed if the contribution of bioenergy to the total energy production is to increase above current levels.

Apparently, willow in short rotation is a cost effective solid biofuel alternative to annual crops, but the actual future yields, landscape planning perspectives, the environmental performance, and landuse flexibility issues needs further consideration.

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Policy Means for Sustainable Energy Scenarios

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Introduction

The global scenarios in "The Limits to Growth" report published in 1972 have been followed by a number of global, regional and national scenarios based on different types of computer models. Recognition of global warming and the danger of climate change have brought scenarios for sustainable energy development into focus. Most of these scenarios have focused on technological solutions. The short-comings of the recent COP15 have emphasized, however, the need for more attention to the non-technical policy conditions, and the potential means for achieving efficient solutions, including activation of private households and changes in life style and economic goals (Meyer *et al.*, 2010).

Consequences of global warming are appearing much faster than assumed just a few years ago and irreversible "tipping points" are few years ahead (IPCC, 2007; Hansen *et al.*, 2008; Kopp *et al.*, 2009). Despite long and tedious preparations for COP15 in December 2009 the final result (Copenhagen Accord, 2009) lacked sufficient concrete commitments for reduction of greenhouse gases (GHGs) after 2012 when the Kyoto Protocol expires.

Human activities in their present form are strongly dependent on the supply of energy. A dominant part of the global energy supply is based on fossil fuels and a dominant part of the climate change is due to emission of CO2 from the use of fossil fuels. For simplicity, this paper focuses on CO2 emission from fossil fuels, but CO2 from deforestation as well as methane (CH4), laughing gas (N2O) and a number of industrial greenhouse gases should be included in a more comprehensive analysis.

The paper will focus mainly on non-technological strategies for mitigation of climate change addressing such questions as national and international equity, limits to growth, population policies, and alternative employment policies. It should be stressed that the focus is on developments in affluent countries with the aim of leaving more environmental space for the less developed countries, where growth in material living standard is often a more pressing and legitimate goal.

Driving forces for climate change

The main factors behind the present climate change may be described by the following simple equation:

$$\mathbf{I = P * A * T,}$$

Where I denotes the Impact on the environment, P is Population, A is Affluence in the sense of general consumption per person, and T is a Technological factor representing the eco-impact per service provided. Growth in any one of these three factors will tend to push upwards the total impact, while a decrease will have the opposite effect. In affluent countries solutions are mainly sought in the T-factor, while the two other factors are ignored or even encouraged to grow. With the acute challenge of climate change it is necessary to consider all three factors.

Population growth

Over the last five decades world population has grown from about three billion to around seven billion. Most of this growth has occurred - and still occurs - in countries with a low CO_2-emission per person. However, in a future where these countries are expected to improve people's general welfare, the number of people will play a significant role in global environmental problems.

Political and religious taboos have often blocked debate on how to handle this population issue, which fortunately is characterized by a large flexibility in options for the long term. According to official UN estimates, world population with no new measures is projected to be around 9 billion by 2050. It turns out that with relatively small changes in number of births per woman, large changes are possible in the long term. For example, with 1.6 or 2.6 births per woman, global population in 2050 could be 8 or 12 billion respectively. The corresponding numbers with these birth rates by 2150 would be 3.6 billion - around half the present population - or 27 billion, respectively (The Population Council, 1998). Sustainability including mitigation of climate change would be a lot easier with the numbers in the lower range of these population scenarios

In many parts of the Western world population is slowly declining. However, government policies in these densely populated, high CO_2 emitting, countries often encourage higher birth rates rather than lower. Most of the global population growth will, nevertheless, take place in the developing world where current energy consumption per capita is much lower than in industrial countries. In recent decades a number of developing countries, especially in the Asian region, have successfully reduced birth rates to around 2.0 or below (UN Population Div., 2008; Mason, 2001). In general this has been associated with a better material standard of living.

Equity

The goal of more equity plays an important role in the quest for sustainable development as it tends to acknowledge and promote economic satiation in affluent societies. Recognition of a world with limited natural resources will tend to make demands for equal right to the use of these resources more morally and politically legitimate. Globally, the lack of understanding of the importance of equity has been demonstrated by the COP15 failure in Copenhagen. The ultimate goal of an economy ought to be human wellbeing in the sense of satisfaction and happiness of the involved human beings. Due to the general observation of diminishing returns of increased income and consumption, equity tends to increase total human wellbeing (Daly, 2007).

Liberalised markets

Commercial markets typically have relatively short time horizons e.g. demanding less than five years pay-back time for investments. In contrast to this, desired radical changes of the energy supply systems require planning horizons of up to 50 years. If this is not taken into account, short-sighted investments based on market competition may block necessary long-term solutions. Investments in new coal plants without *carbon capture and storage* (CCS) and in oil production from tar sand are examples of this. New systems thinking is needed, in some cases requiring that planning and promotion of investments in vital sectors are transferred from the commercial market to government institutions. This applies in particular to an energy sector which has a goal of sustainable development.

Perception of an unlimited world

With few exceptions, economics as a discipline has been dominated by a perception of living in an unlimited world, where resource and pollution problems in one area were solved by moving resources or people to other parts. The very hint of any global limitation as suggested in the report "The Limits to Growth" (Meadows *et al.*, 1972) was met with disbelief and rejection by businesses and most economists. However, this conclusion was mostly based on false premises (Nørgård *et al.*, 2010).

The basic conclusion of the 1972 LtG report was that continuation of the growth policies in population, industrialization, pollution, food production, and the consumption of non-renewable resources would most likely lead to some kind of collapse during the 21th century, due to resource scarcity, over-pollution, over-population, etc.. This catastrophic growth scenario got most attention, but alternative scenarios were also presented in LtG, including one that illustrates that it is possible to change course and reach an environmentally sustainable development path, able to satisfy all people's physical needs. Finally, the LtG report stresses that due to delays in natural and man-made systems, it is essential for achievement of sustainability that global society acts before the environment undergoes irreversible changes and forces undesired changes upon us.

Recent analyses have shown that the developments in the main parameters in LtG have followed quite closely the main trends in the report's standard scenario, which in the model later leads to collapse (van Vuuren, 2009, Turner, 2009). This underlines the fact that the basic structures and aims in world economy have not changed. Thus, it would be wise to pay attention to the analysis of economic structures in "The Limits to Growth" report from 1972 and its two later versions (Meadows *et al.*, 1992; Meadows *et al.*, 2004).

Fear of unemployment

A main argument for continued economic growth is based on the experience that in OECD countries the productivity in the production sector, and to a certain degree also in the service sector, increases by about 2 % per year. Without economic growth this is claimed to lead to more unemployment, overlooking the flexibility of the employment concept. Replacing more consumption by more free time seems like an obvious policy for coping with climate change and other environmental problems. In many European countries, people increasingly prefer shorter paid work time over more income and consumption, in Denmark reaching 73% in 2007 (Nørgård, 2009). This development may be promoted by introducing a general *citizens salary* or *basic income* (Meyer *et al*., 1981).

Personal Carbon Quota

A new scheme for reducing GHG emission is based on Personal Carbon Allowances (PCA) where every adult is allotted an equal, tradable ration of CO2 emission per year related to their consumption of some selected energy services for private households. For simplicity, it is proposed that PCA should be related only to "direct" energy consumption, i.e. energy used for personal travel and for heat and electricity within the household (Fawcett *et al*., 2009).

So far no country has introduced a PCA scheme. The most extensive discussion of PCA has taken place in the UK where UK's *Department for Environment, Food and Rural Affairs,* DEFRA, published a report on the scheme in 2008 (DEFRA, 2008) with a positive evaluation of its potential. Analysis of the potential of PCA in the UK and Denmark (Fawcett *et al.*, 2009) indicates that for these two countries the scheme would address 30 to 50% of the total national emission. The PCA requires systematic government support to the households in order that they may benefit from the scheme and accept it as a positive challenge.

Conclusions

The process of international climate negotiations from Bali to COP15 in Copenhagen has illustrated the need for new and supplementary schemes for mitigation of climate change. This paper has analyzed proposals for new strategic thinking to overcome present barriers and promote efficient mitigation schemes. The main policy recommendations may be summarized as follows:

- New economic paradigm with more attention to sustainability and welfare,
- Efficient population policy,
- More equity globally and within nations,
- Recognition of limits to growth on a finite planet,
- Alternative employment policy with sharing of paid work and more free time.

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Poster Session

The role of EU Structural Funds in the realization of the environmental investments and the effects of the investments in the 2020's final energy consumption in Hungary

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Introduction

The Structural Funds and the Cohesion Fund are the financial instruments of European Union's Regional Policy, which is intended to narrow the development disparities among regions and Member States. Since Hungary is part of the EU it is entitled to a significant amount of subsidy through the Structural Funds to invest into regional development, regional competitiveness and employment, territorial cooperation like cross-border cooperation, transnational cooperation, interregional cooperation. A smaller part of the Structural Funds is to spend on the energy efficiency (EE) and renewable energy (RE) investments. Our aim is (i) to summarize all the subsidies given by the EU Structural Funds so far; (ii) to sum up all the projects in renewable energy utilization and energy efficiency that came into reality since Hungary became a member of the EU; (iii) and to analyse the possibility related to the final subsidy to reach the target proportions (13% renewable by 2020).

Investments in Hungary in the period of 2004-2006

The Environment Protection and Infrastructure Operative Programme (in Hungarian KIOP) was one of the five comprehensive programs of the Hungarian National Development Plan (in Hungarian NFT) for the EU programming period of 2004-2006. The KIOP made the opportunity to invest in three important sectors like environmental protection, energetics and transportation. The head of the Managing Authority was the Ministry of Economy and Transport (in Hungarian GKM), in co-operation with the Ministry of Environment and Water (in Hungarian KvVM). The implementation of the programme was the responsibility of the KIOP Managing Authority (EIMA). The EIMA delegated some of its tasks to Intermediate Bodies. The issue of energy was delegated to the "Energy Centre" Energy Efficiency, Environment and Energy Information Agency Non-Profit Company.

In the 2004-2006 period Hungary got **EUR 3.2 billion** from the Structural Funds. The community sources were ensured from the European Regional Development Fund (ERDF) of the EU Structural Funds (SF). The KIOP's participation in the total budget for this program period was **EUR 440.3 million**, it corresponds to the 13.7% of the financial sources of the NFT. This budget was distributed as it is in Table 1.

Total KIOP budget (EUR)	Environmental Protection (EUR)	Transportation (EUR)	Technical Assistance (EUR)
440.3 million	170 million	251 million	17 million

Table 1.: The KIOP 2004-2006 budget distribution. Source: Investments in the environmental protection and traffic infrastructure in terms of the subsidy of the European Union (in Hungarian), Csilla Csonka, BGF, Budapest, 2007

In 2004-2006, the funds sponsoring for energy efficiency and renewable energies were considered as Environmental Protection investments. They were entitled to use the funds of the "Environmentally Friendly Development of the Energy Management" using **EUR 23 million.** According to this statistics the projects were divided into 3 categories. The proportions are shown in Figure 1.

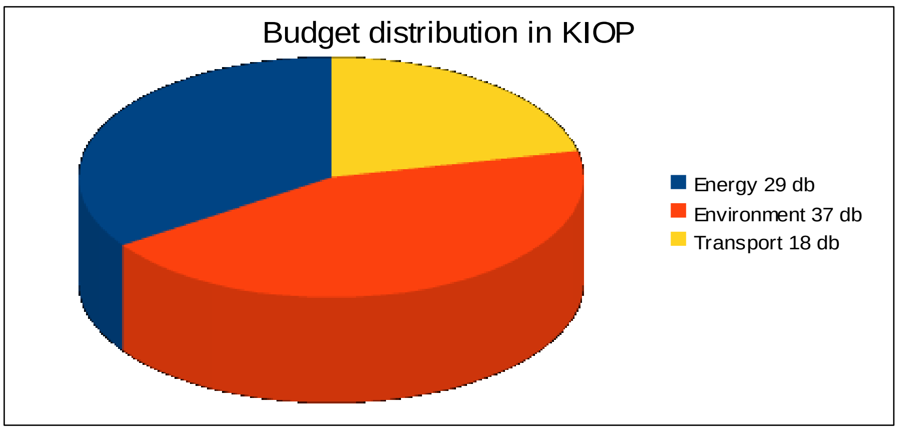

Figure 1.: Budget distribution in KIOP, 2004-2006 Source: GKM

Investments in the Period of 2007-2013

The Environment and Energy Operative Programme (in Hungarian KEOP) is one of the 15 Operative Programmes of the 2007-2013 New Hungarian Development Plan (in Hungarian UMFT), which relates to the EU National Strategic Reference Framework (in Hungarian, NSRK). In this period Hungary gets **EUR 22.4 billion** from the Structural Funds, which is three times more money/year for the development of the projects than in 2004-2006. The Hungarian state gives 15% to this, which amounts to about EUR 26.2 billion. Above that Hungary gets EUR 3.4 billion development source from the Guidance Section of the European Agricultural Guidance and Guarantee Fund (EAGGF) and the Financial Instrument for Fisheries Guidance (FIFG).

From this, **EUR 4.9 billion** is allocated for KEOP including the 15% Hungarian state's co-financing. The renewable energy (RE) partition of the budget is 5.15 % which is EUR 253 million and the energy efficiency (EE) partition is 3.14 % which is EUR 154 million for the whole period.

Vision for 2020

We've found that amongst the energy investments 42% (23 projects) belonged to the Increasing of Public and Industrial Energy Efficiency Program, and 58% (20 projects) to the Renewable Energy Resources Program. In the period of 2007-2013, the 40% of the subsidies paid for the developments in energy efficiency (building energy modernization – such as improvement of heating technology, replacement of the windows and doors, insulations – and lighting modernization) and 60% of the subsidies given so far was used to develop renewable energy utilization.

Based on the outcomes we made an evaluation to see whether we will reach our commitment by 2020, assuming that the intensity of the investments will increase with the present pace until 2020.

Based on the calculation we've found that the share of renewable energies within the final energy consumption in Hungary may reach approximately 94 PJ which is correspond to around 8.6% in 2013 and 134.6 PJ which is correspond to around 12-13% in 2020 which is suit the engagements of the country.

Evaluation of the utilisation of the Structural Funds until 2010

Until 2009 Hungary´s participation from the Structural Funds was EUR 6.696 billion in the new programming period. From this amount of money Hungary received EUR 2.110 billion, which is only the 31.52% of the available subsidy. The unused payment entitlements are EUR 4.586 billion. The utilisation of the Structural Funds is shown in table 2.

Received	Unused	Total until 2009
EUR 2.110 billion	**EUR** 4.586 billion	**EUR** 6.696 billion

2. Table: Utilisation of the subsidy until 2009 Source: http://surjanlaszlo.hu/hu/cikk/197

Table 3. shows below the total amount of subsidy for Hungary and only for KEOP and the utilisation of the money so far.

	Total subsidy (billion EUR)	KEOP (billion EUR)
2007-2013	**22.4**	**4.9**
Until May 2010	**3.5**	**0.145**

3. Table: shows the total amount of subsidy in the KEOP project Source: www.nfu.hu, by author

According to that Hungary has all in all EUR 0.407 billion for the energy investments in the period of 2007-2013, and had already used EUR 0.145 billion which is insignificant compare to the total available. The result of our evaluation shows that if the intensity of the investments will increase with the present pace till 2013 and 2020 this subsidy is enough to reach the 13% in the final energy consumption.

Our opinion is that Hungary should make an effort to spend more money on the energy sector because the amount of the tenderers are growing year by year and Hungary gets the opportunity to increase our final renewable energy supply more than the undertaking 13%. Our natural resources also enable us to develop the renewable energy capacity, as Hungary's attainable socio-economic wind energy potential alone is around 7000 MW by 2020 (Munkácsy, B. 2009).

Overview in some CEE Country

Friends of the Earth Europe and CEE Bankwatch Network are monitoring plans for the use of EU Structural Funds and Cohesion Fund in the energy sector of Central and Eastern European countries over the next seven years.

The beginning of the implementation of the 2007-2013 EU funds coincided with the economic crisis. This situation presented an opportunity for investing in long term developments, in particular by redirecting some of the EU money into sustainable energy investments. Some CEE Member States did react to the economic crisis by redefining their funding priorities and reorganising the OPs. In addition, the Russian-Ukrainian gas crisis of January 2009 struck CEE countries and stressed further the need for enhanced energy security of supply. The countries most hit by the economic crisis in the CEE region first realised the possible win-win effects of energy saving measures for economic recovery and social benefits. They have placed EE/RE projects at the core of national stimulus packages, in which EU funds appear as a central fiscal instrument. The demand for EU funding is therefore increased as a preferred option for member states whose budgets are hit hard by the crisis.

Results of the analyses

The total EU funds for energy efficiency in seven countries (Czech Republic, Hungary, Estonia, Latvia, Lithuania, Poland and Slovakia) are **EUR** 1796 billion between 2007-2013. However, the total amount of EU funds for contracts signed is only EUR 292.43 million which is around 16.3% of the total available subsidy. In the field of renewable energy the situation is even more striking. From the total renewable energy allocations, which account for **EUR** 1751 billion, merely EUR 99.72 million subsidy was contracted. This shows that only 5.7% of the EU funds for renewable energy measures had been absorbed by September 2009.

The analysis showed that while interest and demand in EU funding for energy efficiency and renewable energy measures are on the rise, the same time energy projects are being contracted and spent very slowly. Nearly three years into the 2007-2013 programming period, the number of contracted projects are still low and very little actual spending has been done.

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The CEEH report series (http://www.ceeh.dk/CEEH_Reports) constitutes documentation, validation and scientific results from CEEH. The planned report series consists of eight reports with the following working titles:

1. Description of the CEEH integrated 'Energy-Environment-Health-Cost' modelling framework system
2. CEEH energy system scenarios
3. Assessment of Health-Cost Externalities of Air Pollution at the National Level using the EVA Model System
4. Description and validation of the CEEH-HIA model
5. Demonstration of the full CEEH chain – the HIA line
6. CEEH health impact studies
    a) Description of the CEEH health effects model - selection of concentration-response functions
    b) Laboratory tests of toxicity of combustion particles
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