THESIS FOR THE DEGREE OF LICENTIATE OF PHILOSOPHY

On the Robustness of Air Pollution Policy Cost-Benefit Analysis

STEFAN ÅSTRÖM

Department of Space, Earth and Environment

CHALMERS UNIVERSITY OF TECHNOLOGY

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STEFAN ÅSTRÖM

Department of Space, Earth and Environment
Chalmers University of Technology
SE-412 96 Gothenburg
Sweden
Telephone + 46 (0)31-772 1000

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Abstract

In December 2013 the European Commission proposed an amendment of the National Emissions Ceilings Directive with new ambition levels for harmful emissions of SO_2, NO_x, NH_3, $PM_{2.5}$, and Non-Methane Volatile Organic Compounds. For the first time in European air pollution policy, the proposed ambition levels were based on the future cost efficient emission levels in the EU, as identified by using air pollution policy models based on the standard theories of welfare economics and environmental economics. However, it is not evident that the theory and methods used are robust enough for the results from such models to be converted to policy ambitions. For example, the models are limited by only considering a pre-determined set of end-of-pipe solutions, and by requiring an economic valuation of avoided mortality.

The purpose of the research presented in this thesis was to analyse the robustness of these models. The analysis used different analytical approaches. A cost-benefit analysis was used to identify net socioeconomic benefits of emission control in international shipping. A decomposition analysis was used to test if emission control contributes significantly to emission reductions. In addition, the thesis contains an initial robustness assessment of the foundations of the economic theory used in air pollution policy models.

The results suggest that the robustness of current models would be increased by including options for emission control in international shipping. They also indicate that the current focus on end-of-pipe solutions for control of SO_2 is sufficient for the analysis to be robust. Finally, there are observations and analyses that contradict parts of standard welfare economics and environmental economics but it is yet unclear what these contradictions imply for the robustness of air pollution policy models.

Sammanfattning

I december 2013 föreslog EU-kommissionen att utsläppstaksdirektivet skulle förnyas med nya utsläppsmål för skadliga utsläpp av SO_2, NO_x, NH_3, $PM_{2.5}$ och flyktiga organiska föroreningar. För första gången i Europeisk luftföroreningspolicy baserades de föreslagna utsläppsmålen på den beräknade framtida kostnadseffektiva utsläppsnivån i EU, identifierad genom användning av policy-modeller baserade på ekonomisk standardteori och miljöekonomi. Emellertid är det inte uppenbart att de teorier och metoder som använts är tillräckligt robusta för att resultaten från dessa modeller skall kunna omvandlas direkt till policyförslag. Till exempel så är modellerna begränsade genom att endast beakta befintliga tekniker för direkt utsläppsrening, och genom att kräva att man sätter ett ekonomiskt värde på mortalitet.

Syftet med den forskning som redovisas i denna avhandling var att analysera hur robusta dessa modeller är. Analysen använde olika metoder. En kostnads-nyttoanalys användes för att identifiera socioekonomisk nytta av kontroll av utsläpp från internationell sjöfart. En dekompositionsanalys användes för att testa om direkt utsläppskontroll ger ett signifikant bidrag till utsläppsminskningar. Vidare innehåller avhandlingen en initial bedömning av robustheten i den ekonomiska grundteorin som används i policy-modeller för analys av luftföroreningspolicy.

Resultaten tyder på att resultaten från nuvarande modeller skulle bli mer robusta genom att även beakta kontroll av utsläpp från internationell sjöfart. De indikerar även att nuvarande fokus på endast direkt utsläppskontroll av SO_2 är tillräckligt för att analysen skall vara robust. Slutligen, det finns observationer och analyser som motsäger delar av den ekonomiska standardteorin och miljöekonomi men det är ännu oklart vad dessa motsägelser innebär för robustheten i resultat från policy-modeller för analys av luftföroreningspolicy.

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Appended publications

Paper I: Åström, S., Yaramenka, K., Winnes, H., Fridell, E., Holland, M., The Costs and Benefits of a Nitrogen Emission Control Area in the Baltic and North Seas, submitted to Transportation research part D: Transport and environment

Paper II: Åström, S., Yaramenka, K., Mawdsley, I., Danielsson, H., Grennfelt, P., Gerner, A., Ekvall, T., Ahlgren, E. O. (2017). The impact of Swedish SO_2 policy instruments on SO_2 emissions 1990-2012, Environmental Science and Policy. V. 77. pp: 32-39. doi: 10.1016/j.envsci.2017.07.014

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Glossary of abbreviations used in this thesis, and their meaning

Abbreviation	**Meaning in this thesis**
ARP	Alpha RiskPoll
BC	Black Carbon (sometimes referred to as soot, or elemental carbon), usually considered as a sub-element of $PM_{2.5}$
CAPP	The EC proposal for a Clean Air Policy Package
CBA	Cost-Benefit Analysis
CEA	Cost Effectiveness Analysis
CH_4	Methane
CLE	Current Legislation
CLRTAP	Convention on Long-Range Transboundary Air Pollution
EC	The European Commission
EF	Emission Factor, the amount of emissions that is being emitted due to the combustion of one unit fuel or due to the production of one unit product.
EOP	End-Of-Pipe (used to describe technologies that control emissions primarily through the use of exhaust gas cleaning)
EU	The European Union
GAINS	Greenhouse Gas - Air Pollution Interactions and Synergies
GHG	Greenhouse gases
IAM	Integrated Assessment Model. Air pollution IAMs differ in model setup from the climate IAMs
IIASA	International Institute for Applied Systems Analysis
MTFR	Maximum Technical Feasible Reduction
NECA	Nitrogen Emission Control Area
NH_3	Ammonia
NMVOC	Non-methane volatile organic compounds
NO_x	Nitrogen oxides (NO and NO_2)
OC	Organic Carbon, another sub-element of $PM_{2.5}$
$PM_{2.5}$	Fine particulate matter with an aero-dynamic diameter smaller than 2.5 µm
RAINS	Regional Air Pollution Information and Simulation
SO_2	Sulphur dioxide
TSAP	Thematic Strategy on Air Pollution
VOLY	Value of Life Year Lost. The metric used to value changes in life expectancy due to exposure to air pollution.
VSL	Value of a Statistical Life. The metric used to value mortality rates affected by exposure to air pollution.

Glossary of terms used in this thesis, and their meaning

Term	Meaning in this thesis
Acid deposition	Deposition of acidic components caused by emissions of SO_2, NO_x, and NH_3
Air Convention	The preferred abbreviation of the 1979 UNECE Convention on Long-range Transboundary Air Pollution. Often also referred to as CLRTAP.
Air pollution	Used in this thesis as a summarizing term for emissions of SO_2, NO_x, NH_3, NMVOC and $PM_{2.5}$ (and sub-fractions)
Control costs	In this thesis the term describes the costs for reducing air pollution emissions through the use of end-of-pipe technology, altered production technologies, or other means.
Control option	A specific mean (like EOP technology) available to reduce emissions
Control solution	The combination of options necessary to achieve a certain policy target.
Cost effective (strategy)	Used in this thesis to describe the option or group of options (strategy) that reaches a given emission target at the lowest possible cost.
Cost efficient (solution)	Used in this thesis to describe the air pollution emission level (solution)at which the marginal costs of reducing emissions further is equal to the marginal benefits of the further emission reduction.
Eutrophying deposition	Deposition of eutrophying components caused by emissions of NO_x and NH_3
NEC directive	EU National Emissions Ceilings Directive (Directive 2016/2284/EU on the reduction of national emissions of certain atmospheric pollutants, previously Directive 2001/81/EC)
Net socio-economic benefits	Used in this thesis to describe the total benefits minus the total costs associated with emission control.

Contents

1. Introduction

Emissions of the air pollutants sulphur dioxide (SO_2), nitrogen oxides (NO_x), ammonia (NH_3), non-methane volatile organic compounds (NMVOC) and fine particulate matter with aerodynamic diameter <2.5µm ($PM_{2.5}$) are together and separately causing problems with: human health, acidification, short term climate impacts, eutrophication, vegetation damages, and corrosion. Air pollution is still a concern in Europe and in North America, even though progress has been made.

The World Health Organization (2014a, b) has identified that the largest health risk from environmental causes is mainly driven by human exposure to $PM_{2.5}$ in air. $PM_{2.5}$ in ambient air is mainly constituted of emissions of primary particles ($PM_{2.5}$) as well as of secondary particles (ammonium nitrates & ammonium sulphates) formed in the atmosphere and composed from emitted gases such as NO_x, SO_2, and NH_3. Exposure to $PM_{2.5}$ is associated with premature mortality, heart- and lung-related diseases, and many other illnesses (Thurston, Kipen et al. 2017). In Europe 2012, ~380 000 premature fatalities occurred due to $PM_{2.5}$ in ambient air (Lelieveld, Evans et al. 2015). In Sweden the number of fatalities due to $PM_{2.5}$ exposure is estimated to some 3500 in 2010 (Gustafsson, Forsberg et al. 2014). The latest projections are that air pollution still in 2030 will cause some 260 000 premature fatalities in Europe (Ågren 2016).

SO_2, NO_x and NH_3 emissions might when deposited increase forest soil and fresh water acidification. Sweden is one of the European countries that still suffer from acidification damages. Although recovery is ongoing, 17% of the Swedish water catchment areas are exposed to acid deposition exceeding critical loads for acidification. These 17% are expected to decrease to 10% by 2030 (Fölster, Valinia et al. 2014). Reports are now showing biological recovery in European lakes and streams that were previously uninhabitable due to acidification (Garmo, Skjelkvåle et al. 2014). But still many European countries are projected to experience problems with acidification until at least 2030 (Amann, Borken-Kleefeld et al. 2014).

Emissions of several air pollutants have also been identified to have short term impacts with large regional variation on climate change. Some pollutants, like SO_2, cause cooling, while other cause warming. The air pollutant gaining most attention recently for its impact on climate change is black carbon (BC) which is usually considered a soot sub-fraction of $PM_{2.5}$. BC emissions are considered to have a climate change potential ranging between 120-3200 CO_{2eq}, dependent on climate metric (Myhre, Shindell et al. 2013). Climate change impact has been identified for all of the above presented air pollutants as well as for the effect emissions of methane (CH_4) have on ozone formation. Collectively, these are often termed short-lived climate pollutant (SLCP). Control of SLCP emissions have been shown to enable a reduction in the rate of global warming (Shindell, Kuylenstierna et al. 2012).

In addition to the impacts on human health, acidification, and climate change mentioned above, emissions of the same air pollutants are also associated with several other types of environmental impacts. These will not be covered in detail here but includes eutrophication of soils and surface waters from emissions of NH_3 and NO_x, ozone damages to human health, crops, and ecosystems due to emissions of the ozone precursors NO_x, NMVOC and CH_4, as well as corrosion damages to buildings and materials caused by emissions of SO_2 and ozone precursors. In Europe, the level of concern for these impacts varies. If considering eutrophication current trends and projections show remaining problems in large parts of Europe. The trend for ozone damages is less clear. Results indicate a mixed picture with decreasing peak level concentrations but increasing annual average concentrations. This mixed picture is thought to be due to European emission reductions of ozone precursors (lowering peak concentration) and increased inflow of ozone from other continents in combination with increased CH_4 emissions (increasing average concentrations). Trends for corrosion damages show a steady decline over time (Maas and Grennfelt 2016).

Since emitted air pollutants have residence time ranging between days and weeks in the atmosphere and often travel across nation borders countries need to cooperate in order to effectively reduce negative impacts of air pollution. In Europe, the European Union (EU) thematic strategy on air pollution (TSAP) and the UNECE Convention on Long Range Transboundary Air Pollution (CLRTAP or Air Convention) are the most important policy processes to deal with international agreements on air pollution. The Air Convention was signed in 1979 and have since then implemented eight protocols, out of which the revised 'Multi-Pollutant, Multi-effect' (Gothenburg) protocol is the most recent. This protocol sets country-specific 2020 emission targets for SO_2, NO_x, NH_3, NMVOC, and $PM_{2.5}$. The European Union started a bit later in their efforts to control air pollutants but has today the TSAP and several directives that in various ways regulate EU air quality, most recently the amendment to the National Emissions Ceilings (NEC) Directive (Official Journal of the European

Union 2016). The NEC Directive sets national emission targets for the EU member states and covers the same pollutants as the Gothenburg protocol but with 2030 as a target year.

For years, the protocols and directives have been negotiated with influence from scientific measurements of air quality and environmental indicators as well as model analysis of emission dispersion and policy impact assessments, often under the auspices of the scientific centres of the Air Convention. For the negotiations of new ambition levels, policy impact assessments produced by scientific models have been, and are still, important. These assessments have most often been developed with air pollution Integrated Assessment Models (IAM), such as the UKIAM (Oxley, Apsimon et al. 2003), MERLIN (Reis, Nitter et al. 2003), and RAINS (Amann, Cofala et al. 2004). The RAINS model has later been converted to the GAINS model which in some modes includes greenhouse gas (GHG) control options (Amann, Bertok et al. 2011). Common for all models are that they identify future impacts on emission control costs as well as human health and environmental impacts from proposed policy targets.

The latest European policy strategy proposal, the EC proposal for a Clean Air Policy Package (CAPP), used a new approach to identify an appropriate policy ambition level. Earlier policy proposals and impact assessments were based on ambition levels set by deciding which ambition level for environmental and human health status and control costs that were desired. Models were then used to analyse the impacts of the proposal, including which control options that should be used to ensure the cheapest available control solution (ensuring cost effectiveness). In the new approach a suitable ambition level was instead identified by using models to do a cost-benefit analysis (CBA) of emission reductions and calculate a future cost-efficient (and socio-economic desirable) level of emissions.

Cost effective and cost-efficient are not two words for the same thing. In this thesis the terms ‘cost effective strategy’ and ‘cost efficient solution’ are used to help separate the concepts. In contrast to a cost effective strategy, which identifies the lowest cost to reach a given target, a cost-efficient solution identifies the human health and environmental level at which net socio-economic benefits are maximised. When emissions are at a cost-efficient level the marginal costs of emission reduction are equal to the marginal benefits of emission reduction. This new approach to air pollution policy targets imply that the EC now assumes that it is possible to – with the help of models – identify a range for an optimal future air pollution emission level. This assumption is made despite the fact that the scientific and policy community knows that there are many facts and aspects left outside the current economic analysis. In other words, in 2013 the EC shifted from using models to identify the costs for reaching a given target to using models for identification of the desirable target. In Figure 1 cost-efficient emissions equals an ambition level of 76-92% closure of the gap between the 2025 emission levels in a current air pollution legislation (CLE) scenario and a scenario in which all available control technologies are used (MTFR).

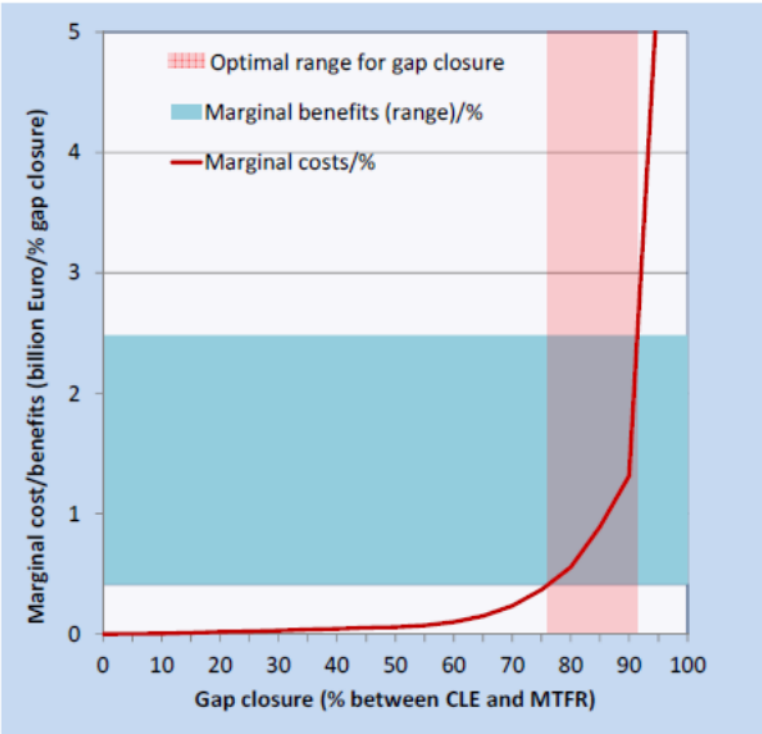

Figure 1: Marginal emission control costs and marginal health benefits in 2025 as estimated in the EC decision support material. Copied from Amann, Borken-Kleefeld et al. (2014)

Prior to proposing CAPP on the 18th December 2013, internal discussions at the EC had lowered the ambition level and the target year had been shifted to 2030, so the proposed 2030 target corresponded to a 67% closure of the 2030 emission gap between CLE and MTFR. During the EU negotiations that followed at the EU council and parliament the policy process continued to reduce the ambition level. In June 2016 the European Union agreed to amend the EU National Emissions Ceilings (NEC) Directive (proposed as a part of CAPP) so that human health impacts in 2030 from air pollution would become 50% of those in 2005. This corresponds to a 40% closure of the gap between CLE control and MTFR control in 2030 given that the objectives of the recently adopted EU climate and energy policy is achieved (54% closure if not considering the EU climate and energy policy).

CBA is a criticised analytical tool (Ackerman and Heinzerling 2005, Ackerman, DeCanio et al. 2009), based on a criticised academic discipline (Schlefer 2012), and it can be questioned whether cost-efficient future emission levels as identified by CBA models are appropriate as basis for target setting in environmental policy. Even when accepting the current CBA concept there might still be approaches in the CBA models that are contentious, and new knowledge from other scientific disciplines that is not yet considered. If altering these approaches or adding this new information the policy message from CBA could change. For example, if considering new knowledge on the links between preterm mortality and exposure to NO_2 (Heroux, Anderson et al. 2015) the relative cost effectiveness of NO_x emission control compared to $PM_{2.5}$ control could increase, and the costs for reaching a given human health target would change. If so, this would thereby affect both the cost effective strategy as well as the cost efficient solution in CBA. Another example is if already considered health aspects (such as human health impacts from exposure to $PM_{2.5}$) would be expanded to also include new knowledge on the types of health impacts from exposure to $PM_{2.5}$ (Thurston, Kipen et al. 2017). Such a change could also alter the policy message from CBA models, but through shifting the cost efficient solution. The ranking of cost effective control options in the cost effective strategy would not necessarily change. The policy message might also change if expanding the number of control options considered in the CBA models. Currently, the CBA models include dedicated air pollution control options from land-based sources. However, there are other types of for example non-technical measures available in reality and international shipping is becoming a more important source of air pollution. Inclusion of new/more control options might change both which options that are considered cost effective as well as which solution that is considered cost efficient.

1.1. Aim and scope of this thesis

The overall aim of this thesis is to examine robustness of the results from current air pollution CBA models. This examination is done in two parts, one applied and one theoretical. The applied part examines robustness within the existing analytical approaches and draws on the results from the two appended papers (methods and findings presented in chapter 2-5).

The hypothesis of relevance for this thesis explored with the research in paper I is that inclusion of more air pollution control options into the current air pollution CBA should motivate more stringent emission levels than what are currently considered as cost-efficient. Paper I thereby adds to with information on whether current air pollution CBA´s are robust with respect to the control options considered. Should analysis in support of policy consider more options than currently available in the model databases? In paper I we analyse costs and benefits of reducing NO_x emissions from international shipping, an emission source currently excluded from the current EU air pollution CBA.

The hypothesis supporting paper II is that air pollution CBA´s that only consider dedicated air pollution control (as is currently the case) severely underestimates the potential for emission reductions. A question related to the robustness of air pollution CBA and partly answered by paper II is: Can it be deemed sufficient to base policy proposals on models that only consider dedicated air pollution control as solution to air pollution problems? In paper II we decompose Swedish SO_2 emission reductions 1990-2012 and identify to what extent it can be claimed that dedicated SO_2 emission control options and dedicated SO_2 policy instruments have contributed to emission reductions.

The theoretical part discusses the robustness of the foundations of CBA. It is based on a separate overview of economic and CBA concepts that might affect CBA results (chapter 6). I present an overview of the setting of standard welfare economics and CBA´s as well as common discussion and critiques towards some of the assumptions in welfare economics and in CBA. I use this overview of welfare economics and the discussion around it as basis for an initial assessment of the robustness of the economic theory supporting air pollution CBA models.

2. Overview of the literature related to paper I and II

Emissions of air pollution from international shipping were for many years regulated to a limited degree. But starting in 2007 emissions of SO_2 became regulated through IMO regulations of the maximum allowed sulphur content in the fuel used. Since then the SO_2 requirements have been strengthened two times and a third is expected by 2020. The regulations are driven by the use of Emission Control Areas, sea regions in which stricter control of emissions is implemented. The Baltic Sea and the North Sea are both emission control areas. NO_x emissions from international shipping are controlled through technology standards which have not yet been ambitious if comparing to requirements on land-based sources. Consequently, NO_x emissions from international shipping in European seas were for a while projected to become larger than land-based emissions (European Environment Agency 2013). To stop this trend, The Netherlands and Denmark as well as the Baltic Marine Environment Protection Commission (HELCOM) proposed to the IMO that the North Sea and the Baltic Sea should be designated as Nitrogen Emission Control Areas (NECA) requiring the strictest technical NO_x emission control standards to be used on new vessels when sailing in these regions.

The emissions, control costs and benefits of implementing a NECA in the Baltic and the North Sea and the English Channel have been partly studied earlier. Kalli, Jalkanen et al. (2013) analysed alternative emission scenarios following international regulations for both the Baltic and North Seas. They found that an implementation of NECA in 2016 would reduce Baltic and North Sea NO_x emissions from 827 to 783 ktonne NO_x in 2020 and from 686 to 183 ktonne in 2040. A report version of the study by Kalli included a sensitivity analysis of an implementation of NECA by 2021, which would reduce NO_x emissions in 2030 by ~25% (from ~830 to ~640 ktonne NO_x in 2030) (Kalli 2013). Campling, Janssen et al. (2013) analysed the cost efficiency of reducing SO_2 and NO_x emissions if a NECA would be implemented by 2016 and found that NO_x emissions in 2030 could be reduced from 202 ktonne NO_x in the Baltic Sea and 503 ktonne NO_x in the North Sea to 108 and 269 ktonne respectively to an total annual cost of 268 million euro/year (800 €/ton NO_x abated). Jonson, Jalkanen et al. (2014) studied impacts of an implementation of NECA in 2016 on emissions and environmental and human health effects. They found that a NECA in the Baltic and North Seas would reduce emissions in 2030 from 293 ktonne NO_x in the Baltic Sea and 642 ktonne in the North Sea to 217 ktonne and 457 ktonne respectively. Finally, Hammingh, Holland et al. (2012) and the Danish Environmental Protection Agency (DEPA) (2012) together analysed emissions, control costs and the environmental and human health effects and benefits of an implementation of a NECA by 2016 in the North Sea. 2030 NO_x emissions in the North Sea would as a consequence of NECA be reduced from 446 ktonne NO_x to 317 ktonne. The total annual costs of reducing NO_x emission would increase with 282 million euro in 2030. Furthermore, they found that benefits would exceed costs by a factor of 2 in 2030 in the main estimates. We could not find any analysis on the net socio-economic benefit of introducing a NECA in both sea regions.

Evaluations of air pollution policies, such as SO_2 instruments, are done with different methods. The most common methods are decomposition analysis, variations of panel data analysis, and case study analysis. In a decomposition analysis, changes in emission levels over time are decomposed into changes over time of the drivers of emissions. Examples of drivers are changes in energy demand, emission control technology use, or structural change (Hoekstra and van der Bergh 2003). Most national- and region-scale decomposition analyses are only loosely connected to individual policies. Examples are Fujii, Managi et al. (2013), Liu and Wang (2013) and Wei, Qiao et al. (2014) which all use different variations of decomposition analysis to identify the main drivers of SO_2 emission reductions in China. Rafaj, Amann et al. (2014a) and Rafaj, Amann et al. (2014b) use decomposition analysis to identify the main drivers of (inter alia) SO_2 emission reductions in Europe. Rafaj, Amann et al. (2014a) show that for EU15, reduced concentration of SO_2 in flue gases (reduced emission factors, EF) were responsible for ~30% of the decoupling of emissions from economic growth and ~50 % of the emission reduction between 2000 and 2010. The decomposition analyses that more directly try to link to individual policies are most often calculated only for one sector and with limited overview of other policies outside the policy studied, such as Hammar and Löfgren (2001), who study the impact of the Swedish sulphur tax on emissions and found that it caused around 59% of the reduction in SO_2 emissions from oil use in manufacturing industries 1989-1995 (59% ≈ 1 ktonne SO_2).

The other two methods are rarely used on a national scale and will hence be covered more briefly here. Panel data analysis typically identifies statistical correlations between emission levels and emission drivers. Examples are Millock and Nauges (2003, 2006) and Hammar and Löfgren (2010). These types of studies are often also sector-specific which prevents national upscaling of results. In a case study analysis, the drivers of emission levels can be analysed both quantitatively and qualitatively and can be closely linked to individual policies (Lindmark and Bergquist 2008, Bergquist, Söderholm et al. 2013). The case study analysis also allows for site-specific and time-specific circumstances to be well represented, as in Bergquist, Söderholm et al. (2013). Another variation of the case study is the policy-directed case study, in which the case is the actual policy institution (Ellerman 2003, Schmalensee and Stavins 2013). As is the case with panel data analysis, the focus of the analysis on a particular sector or industry impedes upscaling of results to national impacts of an individual policy instrument.

For the research questions in paper II we needed to complement the decomposition analysis since it doesn't specify the causality between environmental policy instruments and the driving forces of emissions. In paper II we approached this gap through a qualitative assessment based on a literature overview and mass balance calculations.

This overview indicated a lack of consensus about to what extent SO_2 policy instruments have an impact on the driving forces of emissions. In our analysis these driving forces are: activity levels (fuel use & efficiency); activity shifts (fuel mix changes and changes in products); or emission factors.
Some researchers states that SO_2 policy instruments affect emission factors (Pock 2010, Andersen, Nilsen et al. 2011, Amann, Borken-Kleefeld et al. 2014a). This view can be partly justified by the international policy processes which have focused on the use of Best Available Technologies (BAT), which implicitly promotes the use of end-of-pipe technologies that reduce emission factors (Byrne 2015). Other authors mention the possible but not certain impact of SO_2 policy instrument on fuel-mix changes (Lee and Verma 2000). Earlier rules of thumb were that the capital cost of coal power plant could increase by 25-30% if flue gas desulphurization was added (Das 2006), and such large impact on capital costs could certainly have motivated changes in fuel-mixes from coal to fuels with lower sulphur. This rule of thumb can however be questioned by experiences from ex-post estimates of control costs, which have often showed that the actual abatement costs were lower than the anticipated (Oosterhuis, Monier et al. 2006).

Still other authors – while discussing environmental policies in general – stress the potential combined impact of policy instruments on fuel shifts, energy efficiency and end-of-pipe emission reductions (Kåberger, Holmberg et al. 1994, Xu and Masui 2009, Hammar and Löfgren 2010, Mansikkasalo, Michanek et al. 2011, Rødseth and Romstad 2013). Later analysis also indicates that even command and control instruments such as BAT should be considered to promote innovation and development of manufacturing processes (Lindmark and Bergquist 2008, Bergquist, Söderholm et al.

2013). Innovations in manufacturing processes can often decrease the emissions further than if only SO_2 end-of-pipe criteria requirements are considered.

3. Background

In the 1970´s acidification was recognized as a serious international and transboundary environmental problem which led to the adoption of the Air Convention. Over the years scientific progress helped shape the formulation of the protocols under the Air Convention and the most recent, the Gothenburg protocol, has an effect-based focus where future effects on the environment and human health as well as cost-optimal emission control strategies are identified through the use of models. Effect-based does in this context imply that policy objectives are set for human health and environmental impacts instead of setting objectives for emission levels. The EU efforts to reduce negative impacts of air pollution have developed on a similar path, although often focusing on controlling emissions from specific sectors or fuels. The newer directives, such as the 2001 National Emissions Ceilings (NEC) Directive (amended in 2016) and the 2008 Air Quality Directive, are effect-based and influenced by modelling of environmental and economic impacts of policy proposals.

Emissions of air pollution often stem from the same sources as greenhouse gas emissions and there is thus physical as well as policy links between air pollution and climate change. These links are sometimes reinforcing (co-beneficial) and sometimes antagonistic (causing trade-offs). One typical example of a co-benefit between air pollution and climate change is energy efficiency improvements that reduce emissions of both greenhouse gases and air pollutants, while a typical example of a trade-off is a policy that promotes the use of biofuels, which decrease GHG emissions while risk increasing emissions of some air pollutants. There is however today no international policy that takes an integrated approach and sets emission targets for both air pollutants and greenhouse gases.

3.1. Science used in support of international air pollution policy

Much air pollution research, and specifically policy impact assessments, can be classified as co-production of knowledge between science and policy (Dilling and Lemos 2011). Air pollution research and policy impact assessments have for decades been influenced by policy and vice versa, where the development of the critical loads concept, Integrated Assessment Models such as the GAINS model, as well as effect-based protocols are clear examples (Tuinstra, Hordijk et al. 1999, Tuinstra, Hordijk et al. 2006, Tuinstra 2007, Reis, Grennfelt et al. 2012).

Since the late 1980´s, policy impact assessments have focused on environmental and human health effects as well as emission control costs associated with lower emissions of air pollution (Hordijk and Amann 2007). The number of effects considered has followed the level of advancement in scientific knowledge and the possibility to produce simplified metrics and indicators. Through the development of the 'critical load' indicator (Hettelingh, Posch et al. 1995) the impact assessments can model potential effects on excessive acidification and eutrophication of ecosystems from reduced emission of air pollution. Through the indicators Phytotoxic Ozone Dose (POD) (Emberson, Ashmore et al. 2000), and the accumulated amount of ozone over the threshold value of 40 ppb (AOT40), the effect on ozone damages on vegetation could be assessed, and through progress in materials science links between corrosion damages and air pollution emission levels could be estimated (Tidblad, Grøntoft et al. 2014). In the late 1990´s and early 2000´s the epidemiological knowledge-base was advanced enough (Pope, Burnett et al. 2002) to allow for modelling of human health effects of air pollution. All of these indicators are enabled by regular monitoring of air quality (MSC-West, ccc et al. 2017), experiments and modelling of health and ecosystem impacts from air pollution (Lundbäck, Mills et al. 2009, CCE 2016), as well as research coordination efforts mainly within the Air Convention (Reis, Grennfelt et al. 2012).

Integrated analyses are necessary for policy impact assessments to cover the multiple effects and geographical differences of air pollution. It is also important to use scenario analysis since structural changes in the economy, changes in fuel use, and changes in industrial production all have impacts on emissions. To meet these demands, air pollution IAM´s have been developed. These models build upon the knowledge produced in other research fields, including the indicators presented above. The results from IAM specify which control options that should be used to control emissions and how large the control costs would be for a given target. The results are used to guide policy efforts directed towards international agreements and efforts in certain sectors. Examples of when IAM models have provided direct input to the policy processes are the Gothenburg protocol, (CLRTAP 1999, Amann, Bertok et al. 2011), the European Commission proposal for CAPP (European Commission 2013), and the EU Greenhouse Gas effort sharing decisions (AEA 2012).

Through the progress of the environmental economics discipline, also economic evaluation of air pollution impacts has been made possible. Consequently, also CBA has been used for air pollution policy impact assessments. The decision support material to the Gothenburg protocol as well as the EU Clean Air For Europe (CAFE) programme both used CBA as complementary analysis to verify that proposed ambition levels could be justified from a socio-economic perspective (Holland, Forster et al. 1999, Holland, Watkiss et al. 2005).

The IAM and CBA models used by the Air Convention and within the EU are regularly reviewed with the last major review taking place in 2004-2005 (Grennfelt, Woodfield et al. 2004, Krupnick, Ostro et al. 2005), a smaller internet consultation taking place in 2008, and a review of the epidemiological evidence of health impacts from air pollution in 2013 (WHO 2013a, b).

The policy process partly constrains air pollution policy impact assessments. One such example on how air pollution policy impact assessments adapts to policy realities is through the choice of approaches and methods as well as system boundaries in the analysis. As an example, the air pollution policy impact assessment to the CAPP excluded GHG options that also reduce air pollution from the analysis. This choice of system boundary can be defended by the fact that responsibility for climate policy and air pollution policy in the EU is split between the Directorate-General for Climate (DG-CLIMA) and the Directorate-General for Environment (DG-ENV). DG-ENV cannot propose further GHG control to the EU member states in a process outside the ongoing EU climate policy process. Another constraint is that air pollution policy impact assessments strives to be acceptable to many different types of stakeholders in addition to scientific peers. This implies that state-of-the-art theories, if opaque to laymen, are avoided. But it also implies that black box models are avoided and open access to models and data is promoted.

3.1.1. The most pertinent scientific approaches used to guide air pollution policy

The multi-pollutant, multi-effect approach

The multi-pollutant, multi-effect (MPME) approach ensures that known impacts of air pollutants and their interactions in the atmosphere and the environment is taken into account in analyses. This approach is necessary since several air pollutants affect several impacts, as indicated in Table 1 which describes which connections between air pollution, human health, and environmental impacts that were considered in the GAINS model in 2011 (Amann, Bertok et al. 2011). The approach is also important since several pollutants interact in the atmosphere, including the formation of secondary $PM_{2.5}$ from SO_2, NO_x, and NH_3 as well as the formation of tropospheric ozone through reactions between NO_x and NMVOC & CH_4. The MPME approach thereby emphasise some of the system aspects of air pollution policy.

Table 1: The multi-pollutant/multi-effect approach of the GAINS model (open circles indicate linkages that in 2011 were not yet considered in GAINS). Copied from (Amann, Bertok et al. 2011)

	PM (BC/ OC)	SO_2	NO_x	VOC	NH_3	CO_2	CH_4	N_2O	HFCs PFCs SF_6
Health impacts									
- ambient particulate matter	●	●	●	●	●				
- ground-level ozone			●	●			¤		
Vegetation and ecosystems damage									
- ground-level ozone			●	●			¤		
- acidification		●	●		●				
- eutrophication			●		●				
Climate impacts									
- long-term forcing (GWP100)	●	●	●	●	●	●	●	●	●
- near-term forcing	●	●	¤	¤	●	●	¤		

The impact pathway approach

The IPA (Bickel and Friedrich 2005) builds upon the MPME approach and describes the currently considered appropriate steps of air pollution policy analysis. These steps include modelling of emissions, emission dispersion, environmental & human health impacts, as well as the economic modelling of emission control costs and corresponding economic benefits. It extends the MPME approach by highlighting that air pollution policy also needs to adapt to regional circumstances since population densities and demographics varies over Europe and since the ecosystems of Europe are varying with respect to their sensitivity to deposition of acidifying pollution, eutrophying deposition, and ozone damages. Furthermore, since air pollutants are transported over country borders, and European winds have a general annual average direction, it is also important to know where a potential emission reduction should take place. The impact pathway approach takes all these matters into account and is used as a guidebook for the key analytical steps when doing air pollution policy analysis. An important concept formalised within the IPA is the use of dose-response functions and concentration-response functions. These functions describe in a formalised way the relation between air pollution levels and the impacts on human health and the environment. These functions require input from topic-specific research and are key to analyse the impact of emission changes in policy impact assessments. The ambition level of the analysis sets the boundaries for how meticulous the IPA is done. IPA can use either coupled single-disciplinary models for each step of the analysis or with the use of IAM (CEA) and CBA. CBA will be presented in Chapter 0.

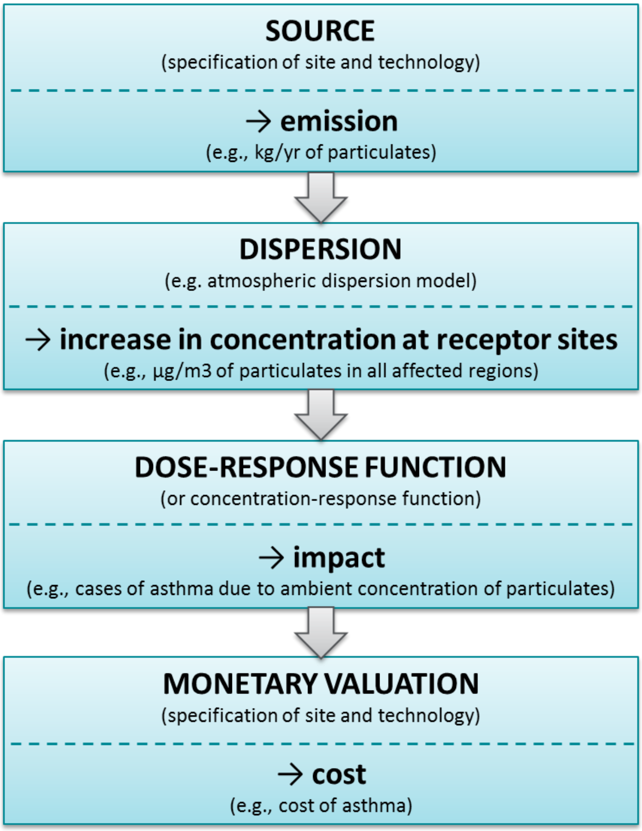

Figure 2: The principal steps of an impact pathway analysis, for the example of air pollution. Adapted from Bickel and Friedrich (2005)

Air pollution integrated assessment modelling

The air pollution IAM that will be discussed in this thesis is the Greenhouse Gas - Air Pollution Interactions and Synergies (GAINS) model, developed by the International Institute for Applied Systems Analysis (IIASA) (Amann, Bertok et al. 2011, Kiesewetter, Borken-Kleefeld et al. 2014, Kiesewetter, Borken-Kleefeld et al. 2015). The GAINS model is developed in many different versions, but the version discussed in this thesis is the European version focusing on control of air pollutants only.

The GAINS model is a bottom up IAM developed to analyse how future air pollution emissions can be reduced to achieve biggest possible positive impacts on the environment and human health to the lowest cost. The model integrates: exogenous scenario data on polluting activities; database information on emission factors and emission control costs; linear form calculations of emission dispersion and deposition over Europe; exogenous data on ecosystem sensitivities and on population demographics; to calculate scenario-specific results on emissions, emission control costs, as well as environmental and human health impacts.

A number of disciplinary models and research feeds in to the GAINS model (Figure 3). Exogenous data on polluting activities is taken either from European scale energy system models and agricultural models such as POLES, CAPRI, and PRIMES (Russ, Ciscar et al. 2009, Britz and Witzke 2014, NTUA 2014), or from national data supplied by national experts. The linear form calculations of emission dispersion is based on calculations with the chemical transport model EMEP (Simpson, Benedictow et al. 2012) and the exogenous data on ecosystem sensitivities is provided by the Co-ordination Centre for Effects (CCE) of the Air Convention (Posch, Slootweg et al. 2012).

To achieve a result with biggest possible positive impact on human health and the environment to the lowest cost the GAINS model minimizes costs for a given policy target. In that respect the GAINS model is used for cost effectiveness analysis (CEA).

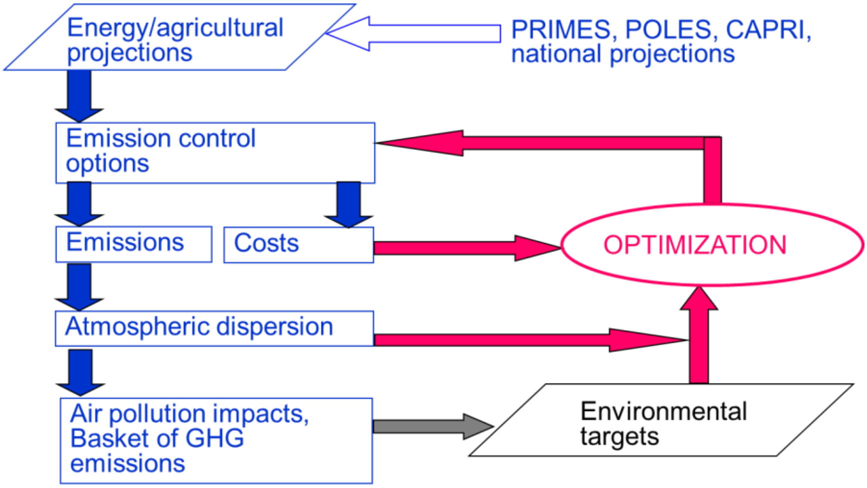

Figure 3: The data and information flow chart for the control cost optimization of the GAINS model. Copied from Amann, Cofala et al. (2004)

Cost effectiveness analysis

CEA is used to identify which control options to use so that a desired target can be met at lowest control cost. Dependent on the model approach used, cost effectiveness can be analysed with different money metrics. In the context of air pollution control costs are expressed as costs associated with the purchase and use of technology, including costs for additional material, waste handling, and sometimes income from by-products. Through inventories of available control options and their control costs these can then be ranked according to their costs so that a cost minimal control strategy can be identified for a given policy target. With the GAINS model, this cost minimal strategy is identified through linear optimization applied to the model setting described above. In short, the minimization uses a policy target on environmental and human health as optimization constraint and then finds the cost minimal solution to reaching that target by varying the use of the available control options. The policy target is based on the gap closure technique by first identifying a baseline emission level and use of control technologies followed by an identification of a maximum technical feasible emission reduction level and corresponding use of control technologies (given application constraints). The policy targets are then introduced as a specification on how much of the gap between the baseline and the maximum that should be closed (Wagner, Heyes et al. 2013).

3.2. Air pollution and climate change

There are close links between anthropogenic air pollution emissions and emissions of greenhouse gases such as CO_2 and CH_4, the two greenhouse gases with highest impact on global warming today (Myhre, Shindell et al. 2013). Combustion of fuel is a main driver of both air pollution and CO_2 emissions, and agricultural activities such as meat production and manure management drives emissions of both NH_3 and CH_4. The relationship between air pollution and climate change can be co-beneficial but also of opposing nature. This is a concern both for physical impacts as well as for impacts from policy initiatives to curb either air pollution or climate change.

3.2.1. Physical interactions between air pollution and climate change

Although the sources of emissions are largely the same, the climate change and air pollution impacts differ in a couple of ways. The residence time in the atmosphere from emitted air pollutants usually range between days and weeks, while emissions of CH_4 has a residence time of roughly a decade and CO_2 atmospheric adjustment time of hundreds of years. Also the impacts differ in terms of time scales. Some air pollution impacts are caused by short term exposure (like acute ozone exposure), and some have an impact that ranges decades (like long-term exposure to $PM_{2.5}$ and acidification). Climate change impacts act on a much longer time scale and through inertia in the global heat circulation system the impacts can last for centuries and more. Closely linked to this difference in time scales are the geographical ranges of the impacts. In general, shorter adjustment time scales implies smaller regional impacts. Air pollution is mainly a local (cities/countries) and regional (continents) problem (although shared by all populated regions of the world), while problems caused by CO_2 and CH_4 are global.

What complicates the matter is that many air pollutants (mainly SO_2, NO_x, PM-fractions, NMVOC) have warming or cooling properties and thereby impact on climate change (Myhre, Shindell et al. 2013). It is mainly sulphur aerosols and fine particulate matter that cause cooling, while some sub-fractions of fine particular matter (BC) as well as tropospheric ozone (affected by emissions of NO_x, NMVOC, and CH_4) is associated with warming. The knowledge about air pollution impacts on climate change is incomplete (Myhre and Samset 2015), impacts have a regional nature (Aamaas, Berntsen et al. 2016), and the climate impacts might be located in other regions than the emission source region (Acosta Navarro, Varma et al. 2016).

One example is the atmospheric brown clouds containing aerosols that have been found to mask the global warming caused by greenhouse gas emissions, but can have both warming and cooling regional impacts, sometimes of larger magnitude than for that of greenhouse gases (Ramanathan and Feng 2009). Conversely, climate change is anticipated to bring about warmer temperatures and changes in precipitation patterns. This might increase tropospheric ozone concentrations in some regions and also change $PM_{2.5}$ concentrations in many regions (likely increasing concentrations in some regions and decreasing in other regions) (von Schneidemesser and Monks 2013). On a global scale, todays' concentration of aerosols (including sulphur compounds) currently counteracts (masks) global warming to an extent corresponding to a radiative forcing (RF) of -0.9 watt / m^2 (CO_2 concentrations cause global warming with an RF of ~1.82 watt / m^2). In contrast, emissions that act as ozone precursors cause an RF of 0.5 watt/m^2. The global average does however hide large regional variation, and the impact of the aerosol components vary (Myhre, Shindell et al. 2013). As an example, Bond, Doherty et al. (2013) find that the direct RF of BC is +0.9 W/m^2, with indirect effects adding more unquantified warming. Again it needs to be stressed that the knowledge is incomplete and the values presented above might be updated.

3.2.2. Interactions between air pollution policies and climate policies

Policies used to control air pollution and climate change implies co-benefits or trade-offs on costs for the economy, as well as on emissions and impacts (Apsimon, Amann et al. 2009). Most available knowledge relate to how climate policies affect air pollution policies. Generally, climate policies are found to be co-beneficial for air pollution, but the size of the co-benefit is largely dependent on the climate policy strategy chosen and how ambitious it is. Studies of co-benefits have also shown that implementation of climate policy alone doesn't lead to achievement of current air pollution policy targets (van Harmelen, Bakker et al. 2002, Rafaj, Schoepp et al. 2013, IIASA 2014).

Earlier studies showed that an expected implementation of the Kyoto protocol in the EU could enable economic co-benefits between air pollution control and GHG emission control by 2010. The size of the economic co-benefit is mainly based on the assumed policy mechanisms. If analysed as separate policies, the reduced costs for air pollution control would amount to 10-20% of total GHG control costs (Syri, Amann et al. 2001). When analysed as integrated policies, the air pollution control costs could be reduced by an amount corresponding to roughly half of the costs for achieving the Kyoto target (van Vuuren, Cofala et al. 2006). Newer studies, analysing economic co-benefits in 2030, also find economic co-benefits of integrating air pollution and climate change policies (McCollum, Krey et al. 2013, Rafaj, Schoepp et al. 2013).

It is not only integration of air pollution and climate policies that are important for economic co-benefits. Also the policy mechanisms used to implement climate policies determine the size of economic co-benefits. For example, analysis show that GHG emissions trading could to some extent reduce European co-benefits between GHG emission control and air pollution control (Syri, Amann et al. 2001, van Vuuren, Cofala et al. 2006, Rypdal, Rive et al. 2007).

By considering the potential for lowering air pollution emissions instead of minimizing air pollution control costs one can study co-benefits and trade-offs on emissions and impacts between air pollution and climate policies. Such studies show that climate policies often lead to co-benefits on environmental and human health impacts due to reduced emissions of air pollutants. For example, if the EU were to strive for a 2 degree climate policy target, this would reduce human health impacts by some 70% compared to a no-climate scenario by 2050 (Schucht, Colette et al. 2015), or by some 35% compared to a Kyoto protocol baseline scenario (Rafaj, Schoepp et al. 2013). Other studies have shown that these types of co-benefits continue to increase until at least 2100 (West, Smith et al. 2013).

The economic and impact-related co-benefits presented above is enabled by the use of energy efficiency measures as well as fuel shifts from solid to liquid, gaseous, and renewable energy. In the long run, also carbon capture and storage may play a role. However, as was the case for economic co-benefits, the climate policy mechanism will have an impact on the size of the co-benefits on emissions and impacts. Air pollution impacts are unevenly distributed geographically, and GHG emissions trading might reduce the co-benefits in Europe. Also, climate policy mechanisms that focus on bio-fuels are at risk of increasing problems with air pollution related human health and also increase warming in the short term through increased used of biofuels in small scale wood combustion which in turn increase emissions of short lived climate pollutants (Rafaj, Schoepp et al. 2013, Åström, Tohka et al. 2013).

In general, while energy efficiency improvements ensures co-benefits between air pollution and climate change, dedicated control of either air pollution or greenhouse gases can cause trade-offs between climate and air pollution objectives. Increased use of biofuels risk increasing emissions of

$PM_{2.5}$, and certain air pollution control technologies are at risk of increasing (to a smaller extent) fuel demand and thereby CO_2 emissions, such as advanced end-of-pipe control in passenger cars (Williams 2012, von Schneidemesser and Monks 2013). The use of diesel cars, implemented in an effort to increase fuel efficiency, is another important example of trade-off between climate and air pollution since diesel cars up until 2017 have been allowed higher $PM_{2.5}$ emissions per kilometre driven. Furthermore, diesel cars have in real life driving been shown to also have large problems achieving the allowed emission limits as compared to gasoline vehicles (Weiss, Bonnel et al. 2012).

The close links between emissions of greenhouse gases and air pollutants has by the scientific community been brought forwards as a rationale for integrating climate and air pollution policy. However, one unexpected effect of these proposals is that they have been taken as an excuse for focusing the policy process entirely on GHG control. This was the case in the negotiations on the NEC directive in the early 2000 (which coincided with the Kyoto Protocol implementation negotiations), and during the first effort to review the NEC directive in 2005-2007 (which coincided with the EU-negotiations for a Climate & Energy package). The same argument was used again in December 2014 when the European Commission suggested to modify the proposal for an amendment of the NEC directive with the motivation that the proposal was: "***To be modified as part of the legislative follow-up to the 2030 Energy and Climate Package."*** (European Commission 2014). In other words, the EC has in three cases appeared to consider dedicated air pollution policy as superfluous since efforts were made to control GHG emissions. This despite the fact that IIASA (2014) show that the latest EU climate & energy package (-40% GHG compared to 1990, 27% renewable energy, 30% improvement of energy efficiency compared to a 2007 baseline projection) would decrease 2030 emissions of air pollutants with only 4-10% compared to the 2030 baseline.

4. Methods

4.1. Cost-benefit analysis

In the versions based on optimization, CBA focus on identifying what a policy target should be to give maximum benefit to society. CBA thereby identifies the cost-efficient solution as in contrast to the cost-effective strategy identified with CEA. In CBA it is presumed that the demand for environmental quality and human health is dependent on the cost of satisfying the demand and that each incremental improvement is worth less than the previous. If this is the case, there is a solution in which the marginal cost for achieving an incremental reduction in emission levels is equal to the marginal benefits of that incremental change. This resulting total emission level is then cost-efficient for society.

CBA can also be used to identify which of the available options (or policies) that would give highest available net socio-economic benefits for society. The results from such a CBA show the ratio of benefit over costs (B/C ratio). If the B/C ratio is above one, the solution gives net socio-economic benefits. This latter version is useful if many control options are available to reach the same target or if the control options studied are non-additive. This type of CBA is the one used in Paper I.

The CBA approach was developed in 19th century France (Pearce 1998). Over the years, CBA practices have been developed by both applied and theoretical researchers. Many guidelines and books have been written on how to do a CBA. In a typical manual, a CBA should include the following steps (adapted from Boardman, Greenberg et al. (2001):

- A specification of the alternatives to be evaluated,
- A decision on whose benefits and costs that should be considered,
- Identification of impacts and how to measure them,
- Prediction of the quantitative change of the impacts,
- Monetization of the changes,
- Discounting of the monetized values if they occur over a period of time and not only in a single year,
- Computing Net Present Value (NPV) of all the alternatives,
- Sensitivity analysis,
- Recommendation on policy action:

The study-specific monetization of the changes is usually prohibitively expensive to analyse and many environmental policy CBAs have come to rely on benefits being assessed with the benefit transfer method. Benefit transfer, '*the use of existing information designed for one specific context to* *address policy questions in another context*' (Desvousges, Johnson et al. 1998) basically implies that either the benefit values or the benefit function from an existing state-of-the-art economic valuation study is transferred to a study on other populations, geographical regions, or policies. The transfer of benefit values can be done through different level of sophistication where the least sophisticated – the direct transfer of values – has been shown to often be the least accurate. Preferably, the transfer of benefit values involves either adjustments for key economic parameters such as GDP per capita and purchase power parity, or studying the trends in values from different studies, or the use of value ranges from prior studies. Transferring benefit functions implies that explanatory variables observable in both the original study and the ongoing study are used to derive a function that explains the benefit value in the original study. The function is then transferred to the ongoing study and used to calculate new benefit values (Johnston, Rolfe et al. 2015).

4.1.1. The CBA method used in paper I

The CBA method used in paper I followed the impact pathway approach. Impact on emissions and control costs were calculated through the compilation of available data and scenarios on emission drivers, emission factors and through the use of technology-specific control cost calculations. Emission dispersion and human exposure calculations were calculated with the GAINS model and monetization of impacts were calculated with the Alpha Risk-Poll (ARP) model (Schucht, Colette et al. 2015).

There are a couple of ways in which the method used in paper I differed from the analytical steps mentioned for a standard approach to CBA. It is not certain that all welfare-relevant economic values, such as existence values and bequest values (Ruijgrok 2004) were considered. With respect to monetization of mortality it appears as if only values related to 'self-sufficiency' are included in the valuation studies such as Desaigues, Ami et al. (2011). Welfare issues like 'concern for good air quality-related health of grand-children' might thereby not be considered in the values used. Furthermore, with respect to benefit transfer it can be said that the CBA in paper I used unit value transfers for all European countries (although using value ranges when literature values were available). One implication is that all Europeans were considered of equal importance. Another implication is that it was assumed that values are equal at all air quality levels and variations of baseline life expectancies (which differ between countries). It should however be mentioned that Desaigues, Ami et al. (2011) present EU average values of avoided mortality. With respect to impacts on morbidity the values used in ARP are derived from health care costs of treating the symptoms and are thereby missing even more welfare-relevant issues than the valuation of mortality.

Instead of using discounting and calculation of net present values, the CBA in paper I calculated annual control costs and benefits and compared these for a target year. By doing so we avoided making a number of assumptions on the temporal distribution of costs and benefits but instead made the assumption that the temporal distribution of costs and benefits would be equally distributed over time. Since a zero discount rate was used and recommended for the benefit estimates of health impacts (Desaigues, Ami et al. 2011) we could avoid the analytical problem of the temporal distribution of health impacts due to reduced long term exposure to air pollution.

With respect to sensitivity analysis it should be mentioned that control cost estimates in paper I used low, mid, high estimates from the literature. The benefit estimates were correspondingly varied following value ranges in the literature on mortality impacts of air pollution exposure. Only one estimate on population projections and central estimates from the literature on human health impacts were considered

Although not necessarily a deviation from the standard CBA approach the choice of population considered in ARP is the populations in the countries affected by the policy proposal. Europeans' potential consideration for 'good health for other European citizens' might thereby not be considered. The valuation studies providing values to ARP does not seem to consider inter-generational or intra-generational justice.

4.2. Decomposition analysis

Decomposition analysis (Hoekstra and van der Bergh 2003) of emissions enables analysis of the relative importance of the driving forces behind emissions and their development over time. The method is considered suitable for analysis of how SO_2 emission reductions are realised (De Bruyn 1997, Stern 2002). Typically, in a decomposition analysis on emissions, chronological data of emission driving forces is collected and used to calculate a baseline emission scenario. Following this, all drivers but one are kept at the base year value and the emission scenario is recalculated. The impact of the driver kept constant is then identified through subtraction of emissions in the recalculated scenario from emissions in the baseline scenario. Re-analysis of historical data is the most common setting for decomposition analysis, but there are examples of decomposition analysis done for future years (Rafaj, Amann et al. 2014).

There are several different types of decomposition analysis. One proposed distinction is between structural decomposition analysis (SDA), using economic input-output data and index decomposition analysis (IDA), using for example energy statistics as data (Hoekstra and van der Bergh 2003). Usually the IDA requires more detailed data, and the driving forces included are often mainly those linked to the physical causality of emissions (fuel use, fuel emission factors etc.). In contrast the SDA through the use of input-output data can show the impact of indirect effects as well as demand effects on emissions. The calculations might be made through the use of econometric models (Stern 2002) or through the use of additive or multiplicative forms of IDA (Hammar and Löfgren 2001, Rafaj, Amann et al. 2014). In some cases the IDA is done purely on indexed values (Divisia index) which allows for comparisons of drivers with different units and not directly physically linked to emissions, such as fuel prices (Hoekstra and van der Bergh 2003, Fujii, Managi et al. 2013). In that respect the Divisia index approach can be seen as a middle step between IDA and SDA.

Our decomposition analysis was based on Rafaj, Amann et al. (2014) and used detailed Swedish energy, industry, and SO_2 emission statistics for 1990-2012 to analyse the relative impacts on SO_2 decoupling from economic growth from structural changes in the overall economy, Fuel use changes (changes in total fuel demand and fuel mixes), changes in industrial productivity, and emission factor changes.

5. Reflections on our results from paper I and II

5.1. Main results

The following conclusions can be drawn from Paper I.

A NECA in both the Baltic and North Seas would give net socio-economic benefits

The results from our analysis are that the 2030 annual control costs would be exceeded by the annual benefits of reduced air pollution problems if a NECA would be implemented in the North Sea or in the North Sea and the Baltic Sea from 2021 and onwards. From our analysis the average benefit-cost ratio would be 5.7 (1.6-12) for the North Sea NECA and 5.2 (1.5-11) for NECA in the North Sea and the Baltic Sea. The benefits are less clear for a NECA in only the Baltic Sea, with an average B/C ratio of 1.5 (0.5-2.9). The most important source of variations in the B/C ratio is the value used for expressing value of avoided mortality.

LNG propulsion gives higher net socio-economic benefits but also larger variance due to climate change impacts

Our results show that a large scale introduction of LNG propulsion engines would give higher net socio-economic benefit but with a larger variation in B/C ratios than for conventional technologies. The B/C ratio in 2030 is 13.5 (0.2-43.1), with 24 out of 27of the calculations giving B/C ratios larger than 1. As for the conventional technologies, these results are sensitive to the value used for expressing value of avoided mortality, but they are also sensitive to the size and economic value of climate change impacts caused by SLCP emission changes and methane slip from the engines. If methane slip would become lower than what is currently estimated, most of the climate change impacts would disappear.

Cost effectiveness of reducing emissions increases with technology utilization

Another interesting aspect excluded from the final version of paper I is how the unit cost of emission control is affected by the number of hours per year that the technology is used. This is an aspect traditionally not focused on when analysing costs of reducing emissions from stationary sources for natural reasons (stationary sources don’t move in and out of emission control areas). This aspect is however relevant for NECA since a fair amount of the ships that would be affected spend a limited time each year in the North Sea and Baltic Sea. In our main analysis we saw that unit control costs were highest for NECA in BAS, followed by NECA in NSE and

BAS+NSE. In a sensitivity analysis we analysed the hypothetical situation in which all the ships would use the control technology for all hours of the year (approximately 5500 hours at sea per year). This situation corresponds to either a lower number of ships taking care of all the transport demand in the sea regions, or that NECA would be implemented world-wide. The average B/C ratio in the sensitivity analysis was 2.9 (1-5.3), 8.3 (2.7-15.2), 6.6 (2.2-12.1) for the Baltic Sea, North Sea, and the Baltic + North Sea respectively.

The following conclusions can be drawn from Paper II.

SO_2 policy instruments explains at least 26-27% of SO_2 emission decoupling from economic growth in Sweden 1990-2012

The decomposition analysis and qualitative literature overview show that SO_2 policy instruments through their impact on fuel-related emission factors and through the installation of a scrubber in one cement production plant was responsible for 26-27% of the decoupling of SO_2 emissions from economic growth. These 26-27% of the decoupling corresponds to 58% of the emission reduction 1990-2012, but are nevertheless underestimations of the impacts of SO_2 policy instruments on SO_2 emissions. Of the other driving forces of decoupling, structural changes explain 43% of the decoupling, fuel mix changes 18%, increased productivity 7%, and confounded emission factor changes 6%.

Only 5-6% of the national decoupling of SO_2 emissions from economic growth can be satisfactory explained by individual SO_2 policy instruments

Identification of individual policy instrument impacts is important if one wants to learn from experience prior to suggesting new or stricter SO_2 policy instruments. Despite the fact that we can identify that at least 26-27% of the decoupling was due to SO_2 policy instruments, it is rarely possible to identify the impact of individual SO_2 policy instruments. With the data and methods available the only non-confounding individual SO_2 policy instruments we can find is the 1996 environmental court ruling to force installation of a scrubber in a cement production plant reduced and the 2007 & 2010 reduction in emission limit values for marine oils. The scrubber installation explains 5-5.8 ktonne (~4% of the decoupling) and the marine oil restriction explains another 0.7-1 ktonne (~1% of the decoupling).

Confounding factors inhibits impact analysis of individual SO_2 policy instruments

The failure to identify effects of most individual SO_2 policy instruments has several explanations. For energy sector emission factor changes the most important explanations are that: 1) many overlapping individual policy instruments were implemented during the period, potentially leading to spill over effects and confounding effects; 2) the typology of fuels and subsectors used in emission inventory in part differs from the classification used in legislative documents; 3) data on age and size of individual plants have not been collected in the emission inventory, neither were fuel price data collected for the fuel classes; 4) information about national legislation procedures (year of inception, announcement, and implementation), local requirements, and investment in SO_2 emission reduction options has not been collected in the emission inventory; 5) there is a lack of knowledge about to what extent industrial actors respond to foreseeable individual instruments before they are implemented.

For the industry sector the analysis was constrained by the available data being aggregated and the lack of accessible compilations of official data on environmental permit decisions. The analysis was also constrained by changes in: 1) size of industrial plants; 2) product assortments; 3) use of recycled materials; and 4) use of process chemicals. All of these co-developed with SO_2 policy instruments and can have had an impact on both emission factors and productivity.

Furthermore, over the period, there were a number of events that could have impacts on national SO_2 emissions but which we had no possibility to include, since they were too diverse and varying for a comprehensive analysis in this paper. Such events include: local voluntary agreements, informative policy instrument initiatives, outcomes from research and development (R&D) policies (Söderholm and Bergquist 2012), and autonomous changes in relative prices of fuels (Schmalensee and Stavins 2013). Neither could we quantify the impact of active engagement in developing international environmental policy.

5.2. Validity of results

5.2.1. Validity of the CBA results (paper I)

Given that the NECA CBA study is based on scenario analysis the results are sensitive to the baseline scenario used. We used previously published estimates on projected transport demand from Kalli, Jalkanen et al. (2013) as basis, but the future transport demand is nevertheless uncertain. However, although a different scenario might change the absolute level of costs and benefits, the ratio between costs and benefits doesn't need to be significantly affected.

Another important part of the scenario analysis is that we assumed that there will be no change in ship vintage as response to NECA (which applies to new ships built after 31st of December 2020). It could be discussed that a policy instrument that only affects technology built after a certain date will lead to a 'building boom' before the date of implementation. In our case this would imply an increase in ship constructions during 2018-2020. Alternatively, one could discuss the risk that ship owners would choose to re-allocate their older vessels to the Baltic and North Seas in response to the NECA. In our study we were not able to take these potential 'announcement effect' dynamics into account due to the low availability of knowledge about the phenomena.

We also assumed that there will be no shift of transport demand from shipping to land based transport as a response to NECA. In other words, we assumed that the NECA would only imply marginal changes to the transport system too small for any modal shifts to occur.

Nevertheless, on a balance, costs and benefits were underestimated. The study did not include the potential for learning effects that would reduce the control costs of emissions. Neither did it include several monetary benefits of emission reductions that would have increased the benefits of emission reductions if included. We therefore consider the results as underestimations of the net socio-economic benefit of NECA in the Baltic Sea and the North Sea.

Finally, when comparing the parts of our CBA NECA results that are comparable to other studies we find that they are relatively well aligned. In our baseline scenario NO_x emissions from the Baltic Sea and North Sea are 748 ktonne NO_x in 2030, which is somewhat lower than in Jonson, Jalkanen et al. (2014) (935 ktonne) and Kalli (2013) (840 ktonne). In our NECA scenario the 2030 NO_x emissions decrease to 554 ktonne (~26% reduction). In Kalli (2013), a NECA would give an emission reduction of 640 ktonne (~25% reduction). Our unit control costs range between ~1440 - ~2800 $€_{2010}$/tonne NO_x dependent on scenario. Campling, Janssen et al. (2013) estimate unit control costs to ~660 $€_{2010}$/tonne NO_x, and DEPA (2012) & Hammingh, Holland et al. (2012) estimate costs to ~890 - ~2910 $€_{2010}$/tonne NO_x. HELCOM (2012) estimate costs to ~1 470-2 060 $€_{2010}$/tonne NO_x.

5.2.2. Validity of the results from the decomposition analysis (paper II)

Data estimates are always uncertain, and some of the reported changes in emissions and emission factors between years can be due to statistical errors. For 2012, the estimated combined uncertainty in national SO_2 emissions is 11%, but ranging between 13 to 74% for individual sectors (Swedish Environmental Protection Agency 2014).

We show that ~32% of decoupling was due to reduced emission factors, which is relatively well aligned with previous studies. Rafaj, Amann et al. (2014) show that SO_2 control gave ~22% of the decoupling of SO_2 emissions from economic growth in western Europe for the period 1960-2010, and Rafaj, Amann et al. (2014) show that SO_2 controls gave ~30% of the decoupling in EU-15 countries for the period 2000-2010. By only considering changes in emission factors as impacts of SO_2 policy instruments we are most likely underestimating the impact of these instruments.

Shortcomings of the decomposition analysis

In the literature – and in paper II – there is little discussion about the fact that it is presumed that the driving forces develop over time independent of each other, in other words the ceteris paribus condition of many economic analyses is allowed to be implemented on chronological and historical data. A typical quote comes from Stern (2002), "*A 1% increase in non-manufacturing industrial output increases sulfur emissions by 0.083% if total output and total energy input and energy mix is held constant.*". The use of ceteris paribus-conditions on historical data is an example of a reductionist approach and further method development is motivated. One clear example is that economic growth is assumed independent of structural changes in the economy. However, to use the ceteris paribus condition to analyse potential future impacts of policy instruments can still be motivated. But in historical data we know the changes that occurred, so to assume away them by forcing the temporal ceteris paribus condition onto the analysis of historical data should reduce the validity of the results from a decomposition analysis.

Furthermore, given that decomposition analysis doesn't specify causality between emission drivers and SO_2 policy instruments, we have to satisfy with the notion that SO_2 policy instruments at least affected emission factors in energy and transport but nothing more, while it is plausible that SO_2 policy instruments had a combined impact on emission driving forces.

Theory and method to analyse this combined impact on historical data still needs to be developed.

Finally, our decomposition analysis was a counterfactual analysis. Many environmental policies, including Swedish SO_2 policies, don't easily allow for the preferable experimental or quasi-experimental counterfactual analysis methods for policy impact evaluations. This is partly due to the national scale of the policies (which omits the use of control groups) but also due to the fact that need for policy impact evaluations isn't considered when designing the policies (Swedish Environmental Protection Agency 1997). Nevertheless counterfactual thinking helps guide the design of decomposition analysis and the identification of potential causal drivers of emissions. Counterfactual analysis is therefore deemed as a necessary tool for environmental policy evaluation to ensure that potential impacts of other confounding factors can be considered when analysing the effect of a policy intervention (Ferraro 2009).

5.3. Scientific contribution to air pollution CBA

With the research presented in this thesis we add to current knowledge mainly in two ways. First we collate the knowledge about options available to reduce air pollution emissions from international shipping. We present new data on emission control costs and analyse the net socio-economic benefits for Europe of reducing emissions from international shipping. The data is clearly presented and easily available to build upon by other researchers. Secondly we analyse to what extent air pollution emission reductions can be considered contingent or independent of air pollution policies and other policy developments. In this work we have also been able to analyse the link between actual policy decisions to actual emission reductions, thereby adding another level of understanding on how effective policy instruments are at reducing emissions.

Of interest for air pollution CBA models is that paper I show that there are more options available with favourable B/C ratios than the options currently considered in the decision support material used by the EC. Paper II shows that it is still relevant to analyse costs of air pollution control in relation to air pollution benefits as stand-alone from other drivers of emission reductions (i.e. 'air pollution control only' scenarios). Even though direct air pollution control is not responsible for all available emission reductions, the impact has been large enough, despite co-existing ambitious CO_2 policies, to support separate analysis and should be considered to still be large enough in the future, especially for countries that have been less ambitious than Sweden in the past.

With respect to robustness of current air pollution CBA the results from paper I indicate that the current approach could add more options in the analysis so as to increase robustness of analysis. With the B/C ratios of NO_x emission reductions found in Paper I it is reasonable to assume that emission reductions from international shipping would be competitive with emission reductions from land-based sources. Including these options should change both the cost effective strategy as well as the cost efficient solution of CBA.

The results from paper II gives an indication that the current consideration of only end-of-pipe emission control options in air pollution CBA models is relatively robust. SO_2 instruments aimed at end-of-pipe emission control were influential in reducing Swedish emissions of SO_2 1990-2012, despite previous large emission reductions, influence from climate policies, as well as a method that reduce the importance of SO_2 end-of-pipe control. Given that Sweden was an early mover on SO_2 control it should be possible to extrapolate this indication to other countries. However, it is not certain that the indications can be extrapolated to other pollutants.

5.4. Implications for air pollution policy

The most important policy-relevant outcome of paper I is that a NECA in only the Baltic Sea wouldn't necessarily give net socio-economic benefits, while it does for the North Sea. The main reasons for this is that many of the countries bordering the North Sea has a higher population density and that the ship traffic runs mainly close to the coast line in the North Sea while it runs in the middle of the Baltic sea as far away from any coast line as possible.

The results from paper I support the recent IMO decision to accept the Baltic and North Seas as NECAs by the 1st of January 2021 (IMO 2017), even though the most cost-efficient solution would have been to go for only a NECA in the North Sea. The relatively clear net socio-economic benefits of introducing a NECA in the Baltic and North seas, and the potential for even larger net benefits through the use of LNG propulsion gives good support to the decision.

Paper II shows that dedicated control of SO_2 still in 1990-2012 was important for the reduction of emissions. This despite the fact that SO_2 emissions already had declined from ~900 ktonne in 1970 (Broström, Grennfelt et al. 1994) to 105 ktonne in 1990 and despite the fact that Sweden in 1991 introduced ambitious climate policies, including a CO_2 tax for specified sectors (increasing from 33 to 110€$_{2005}$ per tonne CO_2 emitted between 1991-2010). This gives support for continued work with air quality policy, in contrast to allowing air pollution policy to be considered only as a part of climate policy. In an international comparison Sweden was early to reduce emissions, and the Swedish energy system was already in 1990 relatively independent from sulphur rich fossil fuels thanks to nuclear power and hydro power. This should imply that European countries that haven't reduced their emissions as much as Sweden and that still have a fossil fuel based energy system should still have a substantial potential to reduce SO_2 emissions through dedicated SO_2 control.

Of further interest is also the fact that dedicated emission control might serve as a safeguard for emission reductions. In our sensitivity analysis we could see that IF the fuel demand and fuel mix of the Swedish energy system had remained as it was in 1990, dedicated emission control would have reduced emissions from these fuels with 45 ktonne instead of the 31 ktonne that would have been the case with a 2012 fuel demand and fuel mix.

6. An overview of the theoretical fundaments of air pollution economics

Earlier in this thesis I have focused on presenting results from research that applies the currently used concept for air pollution CBA´s and discussed how this research affects the robustness of the current concept. In Chapter 6 I move on to discussing the robustness of the theoretical foundations of the current concepts.

The CEA´s and CBA´s of air pollution policies discussed in this thesis are both based on standard welfare economics as described in mainstream text books although rich in technical detail. Air pollution control costs for a specific policy target are calculated by varying the use of existing emission control technologies so as to minimising total control costs. The cost minimization considers technical constraints on applicability of technologies. Unit control costs are defined per control technology and contain information on investments and costs for operation & management, as well as potential impact on resource efficiency. Instead of calculating net present value of the control costs, the annualised cost is estimated taking into account interest rates and technical lifetime of technologies. Benefits of air pollution control are calculated by linking annual physical impacts on primarily human health to monetized values of these impacts. The economic values are derived through economic valuation studies and transferred to a European context through the benefit transfer method. As for the control costs, annualised benefits are calculated instead of net present value. The CBA then allows for identification of a cost-efficient emission level. To avoid over-generalisation of this discussion I will in this text call the economic methods and theories used in air pollution CEA´s and CBA´s presented in this thesis as 'air pollution economics', which to a large extent should be considered as a branch of welfare economics, which in turn is a branch of Economics as taught in mainstream economic text books.

Some of the thinking and assumptions in standard welfare economics and CBA is controversial from an intellectual, ethical, as well as methodological stand-point and therefore deserves special attention in this thesis. I will go through some areas of debate that might affect the robustness of the results in air pollution economics. I am at this stage not able to draw conclusions on whether it is scientifically sound to use CBA results to derive air pollution policy ambition levels, but I hope to shed light on implicit and potentially controversial assumptions that underlies the economic decision support directed towards air pollution policy makers.

6.1. The basic assumptions of standard welfare economics

Economics can be defined as:, "*the study of how societies use scarce resources to produce valuable goods and services and distribute them among different individuals*" (Samuelson and Nordhaus 2010). Welfare economics is the branch of economics in which welfare implications of policies are studied. The standard version of welfare economics is based on a market under perfect competition, in which profit-maximising producers, and utility maximising consumers operate.

The prerequisites that define the market under perfect competition are:

- Trade of homogenous (identical) products and services,
- There are no transaction costs,
- Both buyers and sellers have perfect information (everyone has full knowledge),
- No single actor on the market can affect prices (there are only price takers),
- Actors can enter or exit the market free of charge,
- There is no price discrimination,
- There are no externalities.

The producers and the consumers have in welfare economics been assigned several characteristics. In addition to being profit maximisers (and cost minimizers) the producers experience increasing marginal costs of production and will produce as long as marginal production costs are not higher than the price for the product or service at the market (there is no economics of scale in production). The only way for any producer to increase profits is to be more effective than the other producers.

The consumers choose a bundle of products and services from the market places so that the consumer's utility from consumption is maximised given the consumer's budget constraint. The consumers are characterised by the ability to:

- Compare and rank alternative bundles of products and services (completeness of preferences)
- Hold stable preferences for these bundles (reflexive preferences)
- Have internally consistent preferences (transitive preferences)
- Consider that more is better than less for any product and service, but at declining rate (preferences are strongly monotonic)
- Substitute between different products and services (indifference).

Through these prerequisites and characteristics, trade on the market place for any good or service will lead to a price equal to the marginal cost of production for the product or service, and a welfare maximising equilibrium of price and quantity traded will have been reached.

6.1.1. Common discussions around standard welfare economics

The standard welfare economics' description of the economy has been discussed for years. Some of the discussion has been focused on issues of general importance while other discussion has been based on specific details of the prerequisites and characteristics presented above. I present two discussions of general importance and follow up with some of the discussions around the assumption of the market under perfect competition and the rational consumer. The assumed behaviour of the producer in the market economy will not be specifically discussed.

Is equilibrium analysis always suitable for studies of the economy?

As presented, the models of standard welfare economics assume that the market and the economy strive towards equilibrium, a motion driven by negative feedback mechanisms in the economy. Equilibrium thinking as a way to describe the ideal market origins from the fact that economists started to use equilibrium math, presumably due to the land winnings of equilibrium physics at the time (Beinhocker 2006). The discipline of physics has since then moved on, but much economic analysis has stuck with the assumption that the market strives towards equilibrium. This assumption has been criticized as being too simplistic and not properly representing observed behaviour at the market of for example financial products, or being able to help explain issues such as unemployment, innovation, the emergence of new technologies (which can be affected by positive feedbacks), or transitions in the economy etc. (Beinhocker 2006, Quiggin 2010, Schlefer 2012, Arthur 2014, Stiglitz 2015).

It should also be noted that even if equilibrium would be a valid assumption, game theorists have shown how market rationality in certain cases might provide equilibrium solutions that are not giving the most beneficial solution for society as a whole, such as Nash equilibrium (Nash 1951, The Economist 2016). These studies and proofs are often made in situations that don't share the prerequisites of the market under perfect competition but still deserve attention here for the reason that real life markets also doesn't share many of the prerequisites of the market under perfect competition.

Should one use market rationality in other arenas of decision making?

Despite the fact that at least one of the original thinkers considered economics only to apply to the market place (Mill 1836), much of the current economic analysis assume that rational behaviour at the market can be transferred to other social interactions as well as interactions with nature. Economics is no longer just a science of rational behaviour at the market place, but is now also applied to studying many types of decision making and making policy recommendations in several different arenas. According to Metcalf (2017) this shift of focus was pushed by the work of Friedrich Hayek in the 1930´s and Milton Friedman in the 1970´s. It is today often assumed that the prerequisites of the market and the characteristics of the agents on the market can guide decisions that lead to good management of nature and society. A pertinent example is air pollution economics, which often includes economic valuation of human health and environmental impacts, but also has been used to support creation of markets (such as SO_2 emissions trading) and adjustments of existing markets (such as SO_2 tax).

Is the market under perfect competition too far away from reality?

The market under perfect competition is an idealised model of actual markets. It is clear to most that all the prerequisites are rarely met in reality, and the markets that come closest are the markets for some very basic products such as wheat (Samuelson and Nordhaus 2010). Within academia there is a large body of research exploring several types of exemptions from the basic prerequisites of the market under perfect competition. Some influential areas of research are research on how the equilibrium solution is affected by imperfect information (Akerlof 1970, Stiglitz 2001), the existence of externalities such as air pollution (Ayres and Kneese 1969, Kolstad 2000), or when not all agents are price takers (monopolies, oligopolies, monopsonies, cartels etc.). The real life existence of imperfect information, externalities, and agents having price setting abilities shifts the equilibrium solution of welfare economics. However, in much applied research, most of the basic prerequisites are still used (Gowdy 2004), which in turn could have impacts on policy recommendations from applied research.

The many ways the theoretical consumer differs from a human

The behaviour of the consumer in standard welfare economics has been discussed for decades and with much input from other academic disciplines studying human behaviour, such as psychology. One of the controversies regards the moral of this consumer, who appears to be more egoistic and hedonistic than normal humans. The consumer in welfare economics is supposed to make decisions that maximise his own utility (but might still to an unknown extent derive utility from the wellbeing of others (Pearce 1998)). This controversy has been led on by quotes from influential thinkers like Adam Smith (1776): "*It is not from the benevolence of the butcher, the brewer, or the baker that we expect our dinner, but from their regard to their own interest*" and F.Y. Edgeworth 1881 (quoted in Sen (1977)): "*The first principle of economics is that every agent is actuated only by self-interest*". Experiments have shown that actual behaviour of humans is explained also by other principles than pure self-interest. Examples are concern for fairness (Berg 1995, Engelmann and Strobel 2004) and norms of cooperation (Fischbacher, Gächter et al. 2001, Herrmann, Thöni et al. 2008). The behaviour of humans is also often driven by simplified reasoning (Kahneman, Slovic et al. 1983, Kahneman 2011) and includes several different types of irrational (for the consumer) biases, such as status que bias (Samuelson and Zeckhauser 1988). The utility maximisation of the consumer is also questionable. The prospect theory (Kahneman and Tversky 1979, Tversky and Kahneman 1992) shows (inter alia) how humans evaluate gambles by comparing the outcome with a given reference point rather than comparing it with the total size of the expected reward. It is also noteworthy that the consumer in standard welfare economics appears to have an analytical capacity far beyond any human (Thaler 2000).

6.1.2. Assessing how these discussions might affect the reliability of air pollution economics

Air pollution economics is to a large extent building upon the standard welfare economic definitions of trade at the market place and the behaviour of the consumer and producer as presented in mainstream economic text books. The main exception is that negative externalities cause by air pollution is included in the CBA´s. Therefore, by using results from air pollution economics one takes the underlying assumptions as acceptable for the analysis. But when many of the assumptions of standard welfare economics are questioned it might be the case that the results from air pollution economics are questionable too. I will therefore discuss some potential impacts on air pollution economics based on the overview presented above.

Is air pollution economics in equilibrium?

First of all, air pollution economics relies on equilibrium thinking around the market of air pollution control and the fact that market thinking provides good guidance for air pollution management. Issues that might lead the 'air pollution market' off equilibrium might be new technologies or other ways to reduce emissions, which are rarely included in the analysis. One relatively new technology that appears to be contributing to some parts of a potential large technology transition is the technology used to meet the latest emission standard for control of air pollution from personal cars (Euro 6). Measurements of NO_x emissions from personal cars with diesel engines in actual driving conditions have often shown that emissions from real driving are higher than allowed laboratory values for several Euro standards (Weiss, Bonnel et al. 2012, Lee, Park et al. 2013). In some cases the divergence has been achieved through advanced cheating of the lab tests, as in the Volkswagen Dieselgate. In other cases it has been achieved through optimizing the engine performance so as to fit the lab requirements (cycle beating). The divergence, in combination with Dieselgate, and a corresponding problem with CO_2 (Ntziachristos, Mellios et al. 2014) might have spawned a future technology transition. Several European cities are now considering banning diesel-fuelled personal cars and companies considers stopping investment in research and development on diesel engines. These changes (if implemented) could qualify the market for air pollution control of emissions from vehicles to be characterised by positive rather than negative feedbacks. This would in turn render the standard equilibrium thinking used in current air pollution economics less reliable for this particular market.

The market in air pollution economics

In current air pollution CEA´s it is assumed that a market under perfect competition exists for the control technologies. All countries are assumed to have access to all control technologies at identical level of investment (implying insignificant transaction costs and perfect information). The critique against the perfect information assumption in standard welfare economics might be less pertinent in air pollution economics. The reason for this is that the EU and Air Convention are identifying and documenting technologies and their costs in the Best Available Technology Reference (BAT/BREF) documentation that is part of air pollution policies. However, as in standard welfare economics there is no learning or economics of scale in the CEA´s of air pollution economics. The critique against the ‘no externalities’ assumption is not applicable to air pollution economics since it per definition studies a market for externalities. But part from the efforts to improve the information availability and inclusion of air pollution externalities, the market for clean air in air pollution economics functions as in standard welfare economics.

The producer in air pollution economics

The producers of clean air have the same characteristics as the producers in standard welfare economics. In air pollution economics competition between producers isn't specifically considered but indications of these features are seen in the fact that control costs (investment + operation & management costs) for each technology is fixed, although with slight national variance of operation and management costs dependent on salary levels. The potential producers of clean air are to be found all over society and the optimal choice of control is based purely on cost of the technology. In reality though, time and risk preferences might differ between governments, firms, and private households. These differences might have an impact on the allocation of emission control efforts, an aspect that needs further research.

The consumer in air pollution economics

As in standard welfare economics, the consumer in air pollution economics is able to substitute demand for a service (in this case clean air) for money. The demand function used by the EU for clean air (the horizontal benefit lines in Figure 1) is the same as in the standard welfare economics except for the fact that it (in its current shape) doesn't support strongly monotonic preferences. More research is needed on how thresholds of human health and environmental impacts might affect the demand function, and on how the consumer in air pollution economics differs from humans, and what the implications are of these differences.

6.2. Discussions about CBA

Having established that current air pollution economics accept most of the basics of welfare economics, one can move on to a more CBA-specific discussion. It should be mentioned however, that much of the criticism of CBA does not accept the above presented principles, as in much of the criticism presented by Heinzerling and Ackerman (2002).

To weight costs versus benefits prior to making decisions feels intuitively reasonable and is quite often an activity that many of us engage in in everyday life. A cost-efficient solution, in which the marginal costs of reducing emissions of air pollution are equal to the marginal benefits, is in principle also easy to defend. Any higher effort would cost more than it gives back. In a resource constraint world, ensuring that the policy initiatives with highest cost-efficiency are promoted makes sense. Currently, the main tool available to analyse cost-efficiency of environmental policy is CBA. In the paragraphs below I present the most commonly occurring discussions about and criticisms towards CBA. I start with the discussions that to some extent affects both welfare economics in general and CBA in particular. As with the criticism towards economics in general it is still unclear how much of the criticism towards CBA that is applicable to air pollution CBA.

6.2.1. Main issues of concern for CBA

The challenges with monetization

A CBA involving impacts on goods and services not traded on markets requires monetization of these impacts. Monetization is the practice of identifying our preferences for non-market goods or services, most often through experiments. The ethical foundation and the methods used for monetization are all subject to discussion (Pearce 1998, Frank 2000, Hanley 2001).

One topic for discussion is that the there is a limit to how easily environmental goods and services can be substituted for money, and how such a substitution can be morally justified. As clarifying examples of this controversy one could consider our willingness to accept monetary compensation for the re-instatement of hanging, or disallowing women from voting in public elections (Holland 1996). In short, CBA relies on the questionable ethical stand point that everything can be traded (Hanley 2001) and is sometimes considered as a bit too cynical as discussed by Pearce (1998). Alternatively people might have lexical preferences, i.e. not allowing for substitution between financial costs and degradation of human health and the environment (Pearce 1998), as expressed by Heinzerling and Ackerman (2002) when claiming that a CBA cannot be done due to the sanctity of human life. The existence of lexical preferences might be one reason to the often rather high rate of protest bids in valuation studies.

However, decisions in society sometimes involve decisions on how many lives to save, thereby implicitly pricing the value of avoided mortality (since saving lives often comes at a cost). Examples are considerations of which health care system to have, which roads to build, which safety requirements to put on air planes etc. So even though it can appear morally questionable, economic valuation of human life currently appears to be desirable for social planners.

Another type of ethical criticism is that valuation studies might invoke ones self-interest while people in their day-to-day decisions care for other people as well (Heinzerling and Ackerman 2002). This concern is corroborated in experiments where priming subjects on money creates a context that promote self-oriented thinking. The subjects primed to money behaved in ways that made them "*free of dependency and dependents*" (Vohs, Mead et al. 2006, Bowles 2008). It is however not clarified to what extent altruistic ideas and concern for others are included in monetary evaluation of non-market goods and services (Pearce 1998), but it has been shown that persons with altruistic value orientations assign higher willingness to pay for wildlife preservation than persons with egoistic value orientations (Ojea and Loureiro 2007).

The methods used in valuation studies are often considered problematic (Pearce 1998). One issue for discussion is what valuation studies actually measure. Sagoff (1994) for example argues that the efforts to identify our preferences (through valuation studies) are misguided approximations of utility since preferences not necessarily represents values. A similar discussion relates to the fact that our values for environmental goods and services might be more closely linked to our ideals rather than our consumer preferences. If so, the results from valuation studies (despite their name) might give poor metrics of the value the subjects put on environmental goods and services (Holland 1996, Bowles 2008).

On a more detailed methodological level there are discussions on the subject sample as well as problems with comparing costs and benefits. One example is that most preference studies often are sampled based on Western, Educated, Industrialized, Rich, Democratic (WEIRD) people (i.e. students at European and U.S. universities). The values of WEIRD people are often not representative of the general population (Henrich, Heine et al. 2010). It is also not sure that voting behaviour is similar to real-life purchase behaviour (Heinzerling and Ackerman 2002). Much of this discussion might imply that one of the work horses of environmental valuation, the Contingent Valuation Method (CVM) would give unreliable results. Another issue of concern with the practice of CBA is that valuation seems to be context dependent (value of one week of extra vacation is less sensitive to positional effects than value of extra salary (Frank 2000)). Furthermore, the existence of positional goods – with purchase values not allowing for aggregation into social welfare estimates – causes contextual problems in a CBA if costs of these goods (e.g. a new clean car) are compared with environmental non-positional benefits (e.g. cleaner air) (Jaeger 1995).

It is not only benefits that are difficult to monetize. Experience show that data on costs for emission control often are over-estimated (Jaeger 1995, Heinzerling and Ackerman 2002, Oosterhuis, Monier et al. 2006, Simpson 2014, Chemsec 2015). And projections of future costs require assumptions on inter alia the rate of technical change, an assumption that often is influential to the results (Ackerman, DeCanio et al. 2009).

CBA-specific concerns

There are six main concerns that more directly relates only to CBA, distributional issues, the use of discounting, benefit transfer, treatment of uncertainty, the static nature of most CBA, and undue influence from stakeholders. There are more, but the ones I've excluded are more general in their nature. One example of such excluded critique is the critique claiming that CBA is a black box, a critique shared by most modelling efforts.

Distributional issues

The results of CBA are rarely showing impacts on distribution of costs and benefits (Pearce 1998, Hanley 2001, Heinzerling and Ackerman 2002, Frank 2008). The cost efficient solution is not affected by the distribution of wins and losses, but is satisfied with the fact that there is a potential for the winner to compensate the looser, a position defended by the Kaldor-Hicks potential Pareto improvement criteria.

The use of discounting

Most CBA´s use discounting of future events to enable comparison of costs and benefits occurring today with costs and benefits occurring in the future. Recommended values for the discount rates are often considered as dependent on whether the project is a public or private project, which risk perspective to have, and is often affected by the method used to derive it. Values in the literature range from 0.1% per year (Stern 2006a) up to 9% (Harrison 2010), with the most common values ranging between 3 – 6 % (Moore, Boardman et al. 2004, Godard 2009, Moore, Boardman et al. 2013). The use of discounting, and the discount rate chosen, often has a large impact on the results of a CBA. This is especially evident in CBA´s related to climate change, where impacts several centuries in the future are analysed (Frank 2000, Hanley 2001, Adler 2002). There has been much debate about which discount rate that is suitable for climate change CBA, and the low discount rate in Stern (2006b) was subject to criticism and discussion (Nordhaus 2007, Weitzman 2007, Sterner and Persson 2008). The use of discounting is mainly justified by: the assumption that future generations are expected to be wealthier than current generations; that people (in experiments) express time discounting preferences (pure rate of time preferences); and the fact that money has an opportunity cost (Harrison 2010). However, the use of discounting can also be seen as poor inter-generational justice, since the impacts of future generations is given less importance than current generations. Others, (Heinzerling and Ackerman 2002) discards the use of discounting to do trade-offs between financial and non-financial (read environmental) costs. The currently preferred academic approach to discount rate issues - to a large extent based on experiments and surveys - is to use hyperbolic discounting. Hyperbolic discounting implies rather high discount rates for near term impacts and then continuously lowering them in the future (Weitzman 1998, Grijalva, Lusk et al. 2013). Hyperbolic discounting has it opponents claiming that hyperbolic discounting leads to time inconsistent choices (Laibson 1997, Winkler 2006), a critique that is in turn criticized by others (Hansen 2006). Another proposed solution has been to use lower discount rates for environmental benefits than for financial costs (Horowitz 1996).

Benefit transfer

Most large scale CBAs require the use of meta-data, and monetized values of environmental goods and services are derived through the benefit transfer method (Hanley 2001, Boyle, Parmeter et al. 2013). Benefit transfer might imply risks of over-generalisation and lack of consideration of socio-cultural differences between populations (Hynes, Norton et al. 2012) as well as socio-economic differences.

Uncertainty

CBA is criticized for not sufficiently taking into account the uncertainty of policy outcomes (Hanley 2001), a critique that should be common for many decision support tools. One other type of omitted uncertainty is uncertainty in effects on the considered externalities (like health and environmental impacts). Another important aspect of uncertainty is that many impacts are not monetized at all (Adler 2002, Heinzerling and Ackerman 2002).

Static analysis

CBA is most often static and does not include dynamic effects such as cumulative and indirect environmental effects of policy initiatives (Hanley 2001) as well as market dynamic effects (Heinzerling and Ackerman 2002, Ackerman, DeCanio et al. 2009). Closely related to the critique of CBA being static is the critique of using CBA for analysing policies that leads to large scale changes, which would change the dynamics of the system analysed. One such type of intervention would be ambitious climate policies that might change the structure of society. Environmental economics (a sub-branch of welfare economics) and CBA is by many considered as developed to study marginal changes (the last litre of clean water on the planet is never valued[1]), so the larger the intervention the larger the risk that CBA results

[1] This nuance is one of the key differences between the field of environmental economics and ecological economics. In ecological economics there is valuation of the last litre of clean water (Costanza, R., R. D'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg,

would be invalid. This was one of the criticisms directed against the Stern Review (Stern 2006a), and it also relates to the earlier discussion on equilibrium.

Undue influence from stakeholders on CBA results

CBA has also received critique for not promoting social welfare (Adler 2002). One main driver of this is that some stakeholders can be suspected to have unduly impact on the results from a CBA. Yet again a critique that should be common for most decision support analysis.

6.2.2. Proposed CBA method developments

There are proposals to further develop CBA. Some of the attempts include: Extended CBA (Holland, Hurley et al. 2005), allowing for inclusion of non-quantified aspects, and the very similar Qualitative CBA (van den Bergh 2004) which could enable consideration of the precautionary principle. Other suggestions for developments have been to perform dynamic CBA´s, basically linking more or less advanced equilibrium models to CBA (Kriström and Bonta Bergman 2014), or CBA based on behavioural economics (Gowdy 2004). However, it appears as if the internal theoretical (and philosophical) consistency of these merged concepts still needs to be evaluated.

6.3. Assumptions and limitations in the CBA made for the EC

As mentioned earlier it is still not clarified which parts of the critiques and discussions around economics as a discipline and CBA as a method that is applicable to air pollution CBA. However, there are a number of assumptions and limitations to the air pollution CBA used for CAPP that can be identified.

With respect to the geographical scope of the CBA it is worth noticing that benefits occurring outside the EU are disregarded when identifying cost-efficient emission levels (Holland 2014). Due to the transboundary nature of air pollution the EC initiatives will have positive impacts on the rest of Europe but this is not taken into account in the EC CBA.

S. Naeem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton and M. van den Belt (1997). "The value of the world's ecosystem services and natural capital." Nature **387**, Costanza, R., R. de Groot, P. Sutton, S. van der Ploeg, S. J. Anderson, I. Kubiszewski, S. Farber and R. K. Turner (2014). "Changes in the global value of ecosystem services." Global Environmental Change **26**: 152-158.)

The benefit part of the CBA does not include all of the known impacts of air pollution on human health and the environment. Some of the omitted impacts are air pollution impacts on prevalence of diabetes, skin aging as well as premature birth and gestational weight (Thurston, Kipen et al. 2017). Furthermore, the recent recommendations to include mortality impacts from to exposure to NO_2 (Raaschou-Nielsen, Jovanovic Andersen et al. 2012, Faustini, Rapp et al. 2014, Heroux, Anderson et al. 2015) is not considered in the analysis. Other omitted values are values of ecosystem damages caused by air pollution such as acidification, eutrophication, forest growth damages and reduced biodiversity (Holland, Maas et al. 2015). There are also several control options that are not included in the cost part of the CBA. GHG emission control options are not included (but controlled for in extra scenario analyses (IIASA 2014)). Other omitted options include behavioural changes, scrapping of old technologies, and emission reduction from international shipping.

Instead of calculating net present values, control costs and benefits are annualised. Annualisation implies the assumption that costs and benefits of air pollution control will be homogenous in their distribution over time. The CBA is calculated for a target year instead of being based on calculated net present values of costs and benefits. All costs and benefits are calculated with real monetary values (Currently € at 2005 value). Costs are calculated using a social planner perspective on control costs (4% interest rate on investment, technical lifetime of technology for the annualisation). The assumption of homogenous distribution over time becomes less important since the CBA choose a zero discount rate for the monetary values of human health and ecosystem impacts (Desaigues, Ami et al. 2011).

In the benefit part of the air pollution CBA, the monetary values used to estimate health impacts is based on benefit transfer and is identical regardless of which EU country the impact occurred in (Holland 2014). In this respect, the analysis promotes equality between all EU member states. To ensure compliance with state-of-the-art, guidelines for benefit transfer of the values of non-market goods and services of relevance for air pollution policy has been developed (Pearce 2000). The values on mortality, which dominates the monetized benefits, is partly based on recent analysis presenting EU-average value of life years lost (VOLY) from air pollution. The EU average was based on results from valuation studies in nine EU countries (Desaigues, Ami et al. 2011). However, distributional issues of benefits between rich and poor within a country, or between producers and consumers, or between different sectors are not identified.

With respect to CBA most often being a static analysis it is worth mentioning that in the air pollution CBA used for CAPP, control costs are calculated as static and there is no learning from experience, so costs do not decrease over time or as a consequence of previous investments. Neither does the analysis allow for new technologies or system dynamics. Of relevance for air pollution would be how strategies to improve urban air quality can be intertwined with urban development plans and thereby have impacts on regional economic growth (Whitehead, Simmonds et al. 2006). Furthermore, control costs never disappear. There is always a 'shadow cost/ opportunity cost' of sort for low sulphur fuels for example, despite the fact that high sulphur fuels are no longer on the market in countries like Sweden. Another aspect that deserves mentioning is the fact that costs are usually considered so small that they will not cause any significant impact on economic development. In the analysis the control costs for additional air pollution control are at their max 0.3% of EU GDP in 2030, and ~0.03% for the cost-efficient emission levels (Amann, Borken-Kleefeld et al. 2014).

To analyse potential distributional issues and impacts on economic growth the CBA used for CAPP was complemented with analysis using the GEM-E^3 model (Capros, Van Regemorter et al. 2013). The complementary analysis confirmed the reliability of the assumed low impact on economic development by showing an impact on GDP of ~-0.026% in 2030 compared to the reference scenario from the implementation of the cost-efficient emission level. Furthermore, the complementary analysis also showed small net impacts on employment in the EU (European Commission 2013, Vrontisi, Abrell et al. 2016).

6.4. Other arguments for using CBA as decision support

So far in chapter 6 I have presented common discussion and critique against CBA. However, there are a couple of non-scientific reasons to why CBA is promoted for policy support. Decision makers use economic analysis of policy proposals in general, and CBA of policy proposals in particular because: a) Economics is a discipline that studies humans and societies, which are affected by policy proposals (one of several disciplines); b) Economics and CBA are quantitative, allowing for the production of numbers useful for policy negotiations and targets; and c) Economics is a normative science, which enables decision makers to get an aid in identifying good and bad outcomes (albeit using a hedonistic moral code based on utilitarianism as guide).

In the literature there are also a number of less abstract reasons identified. The primary reason brought up for why CBA can be used is that better options are yet to be identified (Pearce 1998). In other words, results from a CBA should be considered as a best available estimate. It is also the case that the use of CBA is mandated for public sector planning and initiatives in many countries (Pearce 1998, Swedish Road Administration 2015). Also, the ambitions for social change are higher than the financial resources available, making it important to have a way to weight the net benefits of different policy initiatives against each other. Furthermore, striving for cost-efficiency is a prudent way to handle tax payers' money. Yet another reason is that policy initiatives – such as proposals for environmental policies – will have more types of impacts than only environmental, which renders environmental impact assessments insufficient. Another reason sometimes mentioned is that CBA allow for a more democratic decision making than expert opinions (Hanley 2001, Pearce, Atkinson et al. 2006) through the use of willingness to pay studies. This way of thinking can nevertheless be criticised since the interest of minorities might sometimes be more important (Holland 1996).

However, even with these non-scientific arguments for why CBA can be used despite its shortcomings there are some reservations necessary to mention. First and foremost, economists who advocate the use of CBA still emphasise that CBA is preferable to other methods mainly when the policy objective is to achieve cost effectiveness. Furthermore, the same economists also stress the importance of viewing CBA as one of several decision support approaches that should be made prior to decisions.

7. Outlook

With the papers presented in this thesis we have started to analyse whether results from the current approaches to air pollution CBA are robust. In this thesis I have also presented an overview of the fundaments of CBA and discussions surrounding these. In this outlook chapter I therefore present our ongoing studies as well as discuss the potential to add more fundamental knowledge to CBA.

7.1. Our planned and ongoing studies relating to the robustness of current air pollution CBA

In the applied part of the research we are currently analysing to what extent actor perspectives will have impacts on the perceived cost effectiveness of emission control. We compare the social planner perspective with a corporate perspective by altering the interest rate and lifetime of investments used to calculate annual costs of emission control and use the GAINS model to analyse cost effective emission control strategies. These results show if the emission control cost curve in air pollution CBA is robust (if the same control options are used) with respect to actor perspectives.

Secondly we are analysing whether climate metrics used to illustrate the climate impacts of emissions of short lived climate pollutants (SLCP) has an impact on cost effective control of SLCP emissions. We calculate cost effectiveness of SLCP emission reduction for different climate metrics and compare cost effectiveness of the options as a function of climate metric used. These results show whether the emission control cost curve in air pollution CBA is robust with respect to climate metrics chosen.

Third we will compare costs and effects of reducing emissions from land with costs and effects of reducing emissions from international shipping. This study use the GAINS model extended with data from paper I. The results from the study will show if the cost effective pollution control in the Nordic countries would include emission reductions from international shipping if that was added as an option.

Fourth we will analyse to what extent unconventional control options such as non-technical measures and behavioural changes can be added to the portfolio of control options, thereby extending the control cost curve. We also aim to analyse to what extent these measures can be analysed in an IAM framework.

All of the above presented research activities are providing input to the robustness analysis of air pollution economics as it is currently applied. However, as presented in Chapter 6, robustness of the fundamental theory supporting air pollution economics also deserves analysis. For such analysis, the research activities and research questions are currently less clear, and there are many potential research directions.

7.2. Potential ways forward for continuing the development the theoretical foundation in air pollution CBA

The results from models used in air pollution economics of today present a hypothetical and constrained solution to how, where, and by how much emissions of air pollutants should be reduced. The models assume that the agents making investment decisions all base their decisions only on which solution that is cost effective. Furthermore the models assume an international market under perfect competition for emission control technologies and that all agents have perfect information. The available solutions are constrained to an assumed future economy largely unaffected by decisions made to control air pollution, and to established and well defined end-of-pipe emission control technologies that are not subject to any learning effects. The only benefits that matter for the emission levels are benefits that have been monetized and there are no thresholds or decrease in the marginal utility of cleaner air. Given all of the above (and many more scientific aspects) the models present to negotiators a potential solution that is both cost effective and cost efficient for society.

As presented in chapter 6 there is critique against the economic theories that is the basis for air pollution economics and CBA. And there are modelling opportunities that in various ways relax the assumptions and constraints presented above which have not been taken up by the air pollution economics used to deliver decision support to negotiators. During my studies I have had the opportunity to study two alternative approaches to economics: Behavioural economics (Camerer, Loewenstein et al. 2004) and Complexity economics (Arthur 2013, Arthur 2014). Even though I will not present these approaches here, both are descriptive rather than normative and both use more realistic assumptions about how decisions are made than the standard approach to welfare economics and air pollution economics.

Although both Behavioural economics and Complexity economics both fits better with observations and current understanding of decision making it is unclear how these disciplines can help develop air pollution economics further. One of the first things that needs clarification is whether air pollution economics operates in an area of the economy that behaves as it is assumed in mainstream text books. Behavioural and complexity economics are both fields that by some of their proponents are considered to complement the standard theories of economics, not refute them. Camerer, Loewenstein et al. (2004) for example state that the expected utility hypothesis of standard economics "*… is like Newtonian mechanics, which is useful for objects travelling at low velocities but mispredicts at high speeds.*" and that the advancements of behavioural economics "*… does not imply a wholesale rejection of the neoclassical approach to economics based on utility maximization, equilibrium, and efficiency.*". Similarly, Arthur (2014) express that "*… certainly, many parts of the economy could still be treated as approximatively in equilibrium*" and that "*Equilibrium of course will remain a useful first-order approximation, useful for situations in economics that are well-defined, rationalizable, and reasonable static*". A potential overarching question of relevance then becomes: **How do we know (can we know) if air pollution economics belongs to the part of the economy that can be treated as proposed by standard welfare economics and equilibrium thinking?**

If one would be able to answer this question it would give guidance to what the most urgent improvements of air pollution CBA are. Looking at the current focus of behavioural and complexity economics give little guidance. The applications of behavioural economics mostly involve savings, labour economics, and finance. And of relevance for environmental policy is the development of the nudging concept (Sunstein and Thaler 2008) which is (in an environmental context) primarily applied in policies aimed to encourage individuals to make rational decisions on energy use. The applications of complexity economics are rarer. Examples involves finance and technology innovation processes. However, the fact that these applications are not directly related to air pollution economics might just be a result of resource constraints in the research community.

Another overarching question that it would be beneficial to clarify prior to developing new research is: **Why hasn't the state-of-the-art knowledge in economics been taken up (yet) by air pollution economics?** As presented in this thesis, welfare economics have developed fairly advanced approaches to emission control and CBA, while air pollution CBA from an economic perspective might be perceived as rather rudimentary in some of its details. Examples includes that learning is excluded, that only end-of-pipe emission controls are considered, and that marginal benefits of emission control is constant. The answer to this second question would have to consider inter alia the potential impact from active engagement of stakeholders from governments, industrial stakeholders and NGO´s, as well as considering the computational modelling feasibility of air pollution IAM´s.

Until these questions have been answered potential research focus could be to test and discuss the existing proposals for alternative air pollution CBA´s with respect to internal theoretical and philosophical consistency. In this thesis I have presented Extended CBA, Qualitative CBA, Dynamic CBA as well as CBA adjusted to behavioural economics. For policy support, it is also important to keep reminding decision makers that CBA is best used if the policy objective is to achieve cost effectiveness, and that CBA should be one of several types of analysis supporting policy analysis.

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