THESIS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY

# Research and reflections on European air pollution policy support models


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# Abstract

European emissions to air of $SO_2$, $NO_x$, $PM_{2.5}$, $NH_3$ and NMVOC still today cause harm to human health and the environment. These pollutants are associated with the premature death of ~400 000 people annually in the EU (25 000 perished in traffic accidents 2017). Improvements are expected but problems will persist. To abate these effects European countries are engaged in several international agreements, all dependent on interaction between science and policy. For some 15 years scientific decision support to policy-makers was based on integrated assessment models (IAM) combined with quantification of economic benefits in cost-benefit analysis (CBA) of proposed policies. However, in 2013 the European Commission changed approach and used CBA to model socio-economic optimal emission levels and used these levels as basis for a policy proposal. This new approach puts higher demand on model coverage and reliability. It can also be methodologically controversial.

This thesis presents research and reflections on the robustness of air pollution policy support models used by the European Commission, with focus on IAM and CBA. Robustness over climate metrics is analysed with cost-effectiveness analysis of air pollution control options, with sensitivity analysis of metric choice. An $SO_2$ decomposition analysis indicates if consideration of end-of-pipe control options is enough. Robustness of emission control strategies with respect to investment parameters is analysed with IAM, and CBA provides estimates of whether options to reduce emissions from international shipping should be considered in the modelling. Methodological issues are also reviewed.

The results indicate that the models are robust with respect to climate metrics used and the focus on end-of-pipe $SO_2$ options. The modelling of emission control can be sensitive to investment parameters and to the current exclusion of control options on ships. The methodological foundation of CBA is criticised but since environmental policies depend on support also from arenas outside science it remains unclear if CBA-shortcomings impairs the air pollution policy process. Regardless, there are arguments for *inter alia* complementing CBA with analyses based on non-economic decision rationales. Finally, the thesis provides insights and suggestions for air pollution policy modelling and research that should be considered in the future.

**Keywords**
Air pollution, Short-lived Climate Pollutants, Cost-benefit analysis, Integrated assessment modelling, Decomposition analysis, Policy support modelling, Climate metrics, International shipping, Economic perspectives, Economic methodology

## List of publications

This thesis is based on the work contained in the following papers, referred to by roman numerals in the text:

I. Åström S. & Johansson D., Forthcoming, The choice of climate metric is of limited importance when ranking options for abatement of near-term climate forcers, Climatic Change, DOI: 10.1007/s10584-019-02427-4
*SÅ had the idea, designed the study, collected data, and made the analysis. SÅ and DJ interpreted results. SÅ wrote the paper with comments and additions from DJ.*

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, November 2017, pp: 32-39, DOI: 10.1016/j.envsci.2017.07.014
*SÅ had the idea and designed the study. KY, IM, HD, AG, collected data. SÅ made the analysis with contributions from AG. SÅ interpreted results with contribution from PG, TE, EA. SÅ wrote the paper with comments from PG, TE, and EA.*

III. Åström, S., Kiesewetter, G., Schöpp, W. Sander, R., Andersson, S., 2018, Investment perspectives on costs for air pollution control affect the optimal use of emission control measures, Clean Technologies and Environmental Policy, DOI: 10.1007/s10098-018-1658-4
*SÅ had the idea and designed the study. GK, WS, RS, and SA developed the model, with minor contribution from SÅ. GK, WS, and RS collected data. SÅ made the analysis with contributions from SA. SÅ interpreted results. SÅ wrote the paper with helpful comments from GK.*

IV. Åström, S., Yaramenka, K., Winnes, H., Fridell, E., Holland, M., 2018, The Costs and Benefits of a Nitrogen Emission Control Area in the Baltic and North Seas, Transportation research part D: Transport and environment, V. 59, March 2018, pp: 223-236, DOI: 10.1016/j.trd.2017.12.014
*SÅ had the idea. SÅ designed the study with contributions from HW, EF, and MH. SÅ, KY, WH, and MH collected data. SÅ made the analysis with contributions from KY and HW. SÅ interpreted results and wrote the paper with helpful comments from KY, HW, EF, and MH.*

V. Åström S., Manuscript, A methodological reflection on the European Commissions' 2013 use of CBA for air pollution policy,

## Other relevant publications by the author


A. Rypdal, K., Rive, N., Åström, S., Karvosenoja, N., Aunan, K., Bak, J., Kupiainen, K., Kukkonen, J. (2007). Nordic air quality co-benefits from European post-2012 climate policies, Energy Policy. V. 35. pp: 6309-6322. DOI: 10.1016/j.enpol.2007.07.022

B. Apsimon, H., Amann, M., Åström, S., Oxley, T. (2009). Synergies in addressing air quality and climate change. Climate Policy. V. 9. Iss. 6. pp: 669-680. DOI:10.3763/cpol.2009.0678

C. Åström, S., Tohka, A., Bak, J., Lindblad, M., Arnell, J., 2013, Potential impact on air pollution from ambitious national $CO_2$ emission abatement strategies in the Nordic countries – environmental links between the UNFCCC and the UNECE – CLRTAP, Energy Policy 53, February 2013, pp: 114-124, DOI: 10.1016/j.enpol.2012.10.075

D. Munthe, J., et al. (2016). Klimatförändringen och miljömål., http://www.naturvardsverket.se/Documents/publikationer6400/978-91-620-6705-2.pdf?pid=17304

E. Grennfelt, P., et al. (2017). Forskning för renare luft., www.scac.se

F. Maas, R. and P. Grennfelt (eds). (2016). Towards Cleaner Air - Scientific Assessment Report 2016. Oslo, EMEP Steering Body and Working Group on Effects of the Convention on Long-Range Transboundary Air Pollution

## Populärvetenskaplig sammanfattning till Erik Meuller och andra lärda lekmän

Denna avhandling sammanfattar min forskning och mina reflektioner kring de datormodeller som används för att ge direkt stöd till framtagande av internationella luftföroreningsavtal. Mer specifikt har det främsta syftet med forskningen varit att studera pålitligheten i de policyrekommendationer som ges från främst kostnadsnyttoanalyser av åtgärder för att minska utsläpp av luftföroreningar. Men först ut i denna sammanfattning gäller det att förklara omfattning och begrepp, och jag börjar med det enklaste först.

De luftföroreningar som är aktuella i denna forskning är gaserna svaveldioxider, kväveoxider, ammoniak, lätt-omvandlade (volatila) organiska kolväten, samt små fasta partiklar med en diameter på mindre än 2,5 mikrometer (ett hårstrå är ca 20–100 mikrometer i diameter). När det är tillämpligt inkluderas även utsläpp av växthusgaserna koldioxid och metan i analyserna. Dessa luftföroreningar kan sprida sig mycket långt i atmosfären och orsakar tillsammans och var för sig ett flertal effekter på miljö och hälsa. Mest uppmärksammat är försurning, övergödning, för tidiga dödsfall, samt växtskador orsakat av höga halter marknära ozon. Som exempel kan nämnas att luftföroreningar ligger till grund för cirka 400 000 dödsfall i EU genom påverkan på främst hjärta och lungor, medan olyckor i trafiken orsakar cirka 25 000 dödsfall. På grund av att luftföroreningar sprider sig över nationsgränser behöver utsläppen regleras i internationella miljöavtal. Vilken effekt, och hur stor effekt det blir av ett utsläpp är beroende på var och när utsläppet sker. Det är dessutom så att möjligheten att minska utsläpp varierar beroende på länders ekonomiska utvecklingsnivå samt industriell struktur. Att förstå effekten av en utsläppsändring är helt enkelt svårt, men vi vet att internationella miljöavtal baserade på lika stor utsläppsminskning i alla länder riskerar leda till insatser som är samhällsekonomiskt ineffektiva.

För att kunna ge en bild av vilka effekter en given utsläppsminskning kan ge har det internationella forskarsamfundet sedan 80-talet samverkat för att skapa integrerade bedömningsmodeller som stöd till framtagande av internationella miljöavtal. Dessa modeller kan ses som långa sammankopplade ekvationskedjor som i olika steg matas med data om exempelvis naturmiljön, samhällets bränsleanvändning och ekonomisk aktivitet, utsläpp, utsläppsspridning, befolkningsstorlek, tillgängliga åtgärder med mera. Dagens versioner har möjlighet att göra scenarier för framtida luftkvalitet i samtliga europeiska länder och kan till och med räkna ut det billigaste sättet att nå ett givet mål för luftkvalitet, inom vissa tekniska gränser, och även räkna ut förväntad luftkvalitet i en stor del av Europas städer. I början på 90-talet började man på ett strukturerat sätt även räkna pengavärden på miljön och människors hälsa, och koppla dessa värden till förändringar i luftkvalitet. Från en sådan koppling kan man få fram ett ekonomiskt (monetärt) värde av en utsläppsändring. Om man sätter ihop denna monetära värdering med resultat från den ovan nämnda integrerade beslutstödsmodellen kan man beräkna kostnader och nyttor av förslag på utsläppsminskningar, det vill säga göra en kostnadsnyttoanalys. Både integrerad bedömningsmodellering och kostnadsnyttoanalys (och även den mindre komplicerade dekompositionsanalysen) är exempel på beslutstödsmodellering och står i fokus för min forskning och mina reflektioner.

Denna typ av modeller har använts länge för att bedöma om diverse förslag på utsläppsminskningar är försvarbara för samhällsekonomin, och forskarsamfundet samt beslutsfattare är bekanta med modelltypens brister och förtjänster. Men i december 2013 bytte EU-kommissionen det sätt på vilket beslutstödsmodellering användes. EU-kommissionen presenterade då policyförslag för utsläppsminskningar framtagna till stor del med hjälp av modellering - modellen hade satts i förarsätet istället för beslutsfattare. Man gick från att beräkna det mest kostnadseffektiva sättet att nå ett givet mål till att beräkna vilket mål som skulle maximera välfärdseffekter - där välfärd beräknas som besparade pengar att spendera på andra saker. Detta skifte är värt att diskutera då det kräver hög modellprecision och baseras på en ifrågasatt metodik - bland annat ifrågasätts lämpligheten och möjligheten att sätta ett pengavärde på människoliv och natur.

I min forskning har jag använt mig av samma beslutstödsmodellering som EU-kommissionen använder, om än med snävare geografisk avgränsning. Den röda tråden i forskningen är att den undersöker om modelleringen är stabil med avseende på några av de parametrar som ingår i beräkningarna, och om modelleringskoncepten ger robusta rekommendationer till beslutsfattare. Urvalet av parametrar utgår från ett ekonomiskt perspektiv. Mer specifikt kan sägas att den del av forskningen som inriktar sig på själva metoden studerar fyra parametrar.

1) Kommer det delvis normativa valet av indikator för att representera klimatpåverkan från luftföroreningsutsläpp påverka vilka åtgärder som anses kostnadseffektiva?

2) Är modelleringens fokus på utsläppsrenande tekniker tillräckligt omfattande för att anses representativt för luftföroreningspolicy?

3) Påverkar valet av ekonomiskt perspektiv i modelleringen de tekniker modelleringen anger som mest kostnadseffektiva?

4) Skulle ett inkluderande av åtgärder riktade mot utsläpp från internationell sjöfart ge möjlighet till ännu mer kostnadseffektiv utsläppsminskning?

Resultaten från dessa analyser visar att modelleringen är tillräckligt tillförlitlig vad gäller val av klimatindikator för beräkning av mest kostnadseffektiva åtgärder. Till exempel är det så att oavsett om indikatorn ’Global klimatförändringspotential’ över 100 år ($GWP_{100}$) eller ’Global temperaturhöjningspotential’ om 20 år ($GTP_{20}$) används kommer den stora majoriteten av beräkningar för Sverige resultera i att exempelvis ett skifte från vedanvändning till pelletsanvändning för uppvärmning är mer kostnadseffektivt än att täcka över flytgödselbrunnar. Denna robusthet gäller dock inte vid beräkning av välfärdsmaximerande utsläppsminskning. Modelleringen kan även anses robust i dess fokus på utsläppsrenande åtgärder, i alla fall för utsläpp av svaveldioxid. Resultaten av den andra analysen visar att cirka 48 procent av den svenska utsläppsminskningen av svaveldioxid 1990–2012 utgjordes av utsläppsrenande åtgärder - en hög siffra givet att Sverige redan år 1990 kraftigt hade minskat sina utsläpp från 1970 års toppnotering. Då Sverige ligger före andra länder vad gäller utsläppsminskning av svaveldioxid bör det således finnas gott om utrymme för ökad användning av reningsteknik i andra länder. Däremot riskerar valet av ekonomiskt perspektiv vid beräkningarna påverka vilka tekniker som anses kostnadseffektiva, och framtida analyser bör därför kontrollera detta innan rekommendationer till beslutsfattare ges. Det vore även bra att i fortsättningen låta beslutstödsmodelleringen inkludera utsläppsminskande åtgärder i den internationella sjöfarten.

Den ekonomiska metodik som ligger till grund för kostnadsnyttoanalys är däremot mer problematisk. Det ramverk som ligger till grund för kostnadsnyttoanalys har lite stöd i systemvetenskapliga teorier, vilka snarare visar hur det vi idag vet om ekonomisk utveckling innebär att det finns många olika framtidsbilder (alla rimliga) för framtida utsläpp redan innan effekten av en luftvårdspolicy studeras. Att ur denna mångfald av framtidsbilder analysera en effekt av en luftvårdspolicy blir därmed omöjligt om full hänsyn tas till det ekonomiska systemets komplexitet. Framtiden är med andra ord genuint osäker och en jämförelse mellan en framtid med policy och en utan policy blir precis lika osäker och otydlig. Dessutom, ekonomisk rationalitet såsom den brukar beskrivas i kostnadsnyttoanalys (asocial, köpslående, självcentrerad, nyttomaximerande), har sedan årtionden visats kunna leda till beslutsförslag som går på tvären mot dagens norm kring rationella beslut inom miljöområdet (social, hänsynsfull, hållbar). Vidare är det så att psykologiska och ekonomiska experiment kunnat visa upp fler och fler exempel på när individers och gruppers beslutsfattande skiljer sig från ekonomiskt rationellt beslutsfattande, vilket i sin tur innebär att en kostnadsnyttoanalys av ett policyförslag i praktiken riskerar vara en dålig representation av påverkan på individers välfärd. All denna kritik är giltig kritik mot den kostnadsnyttoanalys som EU-kommissionen lät använda för att ta fram ett förslag på ny luftvårdspolicy i EU.

Men denna kritik måste vägas mot andra viktiga aspekter. För det första, när bästa tillgängliga teorier visar en framtid som kan bli lite hur som helst behövs ytterligare antaganden och förenklingar för att kunna ge stöd från vetenskapen till beslutsfattare. På så sätt är en kostnadsnyttoanalys, även om den inte är den bästa representationen av en framtida verklighet, användbar då antaganden och teori sedan länge har blivit klargjorda och diskuterade. Den utgår dessutom från ett teoribygge som EU sedan många år haft som ledstjärna: kostnadseffektivitet. För det andra måste man sätta in beslutstödsmodelleringen i sitt sammanhang som en kugge i ett stort policymaskineri. En genomgång av den nästan treåriga policyprocess som följde på EU-kommissionens förslag antyder att beslutstödsmodelleringen främst är en startpunkt för förhandlingar. Aspekter som modelleringen inte tar hänsyn till kan bli representerade under påföljande förhandlingar. Till sist, ekonomiska konsekvensanalyser är inskrivna i EU-lagstiftning, och ovanstående kritik innebär inte att kostnadsnyttoanalyser bör skrotas. Den vetenskapliga processen möjliggör ju att beslut slipper grundas på förutsägelser baserade på någon som har spått i tenn, kastat tärning, lekt expert, eller känt vartåt vinden blåser. EU-kommissionens skifte innebar trots allt en utökad användning av tillgänglig kunskap och analyskapacitet för att ge stöd till beslutsfattare.

Men det finns fortfarande alternativ till det angreppssätt som användes. Det första och mest självklara är att forskarsamfundet måste fortsätta sina ansträngningar att ta fram en metodik som är mer realistisk och samtidigt klarar av att behålla logisk stringens och tydlighet. Det måste även tas fram en modellering som kan visa hur redan uppsatta politiska mål för miljö och hälsa kan nås i en snar framtid. I väntan på detta kan forskarsamfundet bidra med analyser till beslutsfattare baserade på flera typer av rationalitet än endast ekonomisk. Även om kostnadsnyttoanalys i fortsättningen kommer bidra med analyser av socio-ekonomiskt önskvärda utsläppsnivåer så bör det kompletteras med analyser som visar vilka utsläppsnivåer som skulle kunna säkerställa att uppställda mål nås, samt med analyser av vilka utsläppsnivåer som ger en jämnare fördelning avseende åtgärdskostnader och/eller miljökvalitet. Denna typ av kompletterande analyser ger beslutsfattare ett större underlag som bättre svarar mot olika politiska prioriteringar, samtidigt som pålitligheten i teori och modeller tillåts ligga inom rimliga gränser.

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## Glossary of abbreviations used in this thesis

| Abbreviation | Meaning in this thesis |
|---|---|
| AOT40 | Accumulated amount of ozone exposure over 40 parts per billion |
| AP | Air pollution |
| AP-CBA | Air pollution cost-benefit analysis |
| AP-IAM | Air pollution integrated assessment model |
| ARP | Alpha RiskPoll |
| B/C ratio | Benefit-cost ratio |
| BC | Black Carbon (sometimes referred to as soot, or elemental carbon), a sub-element of $PM_{2.5}$ |
| CAFE | Clean Air for Europe |
| 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 (Air Convention) |
| $CO_2$ | Carbon dioxide |
| DG-CLIMA | European Commissions Directorate-General for climate |
| DG-ENV | European Commissions Directorate-General for environment |
| EC | The European Commission |
| EU | The European Union |
| GAINS | Greenhouse Gas - Air Pollution Interactions and Synergies (model) |
| 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 |
| LNG | Liquid natural gas |
| MC | Marginal cost of emission control |
| MTFR | Maximum Technical Feasible Reduction |
| NEC | National Emission Ceilings |
| NECA | Nitrogen Emission Control Area |
| $NH_3$ | Ammonia |
| NMVOC | Non-methane volatile organic compounds |
| $NO_x$ | Nitrogen oxides (NO and NO2) |
| NPV | Net present value |
| OC | Organic carbon |
| $PM_{2.5}$ | Fine particulate matter with an aero-dynamic diameter smaller than 2.5 µm |
| POD | Phytotoxic ozone dose |
| RAINS | Regional Air Pollution Information and Simulation (model) |
| RF | Radiative forcing |
| SCR | Selective catalytic reduction |
| SDG | Sustainable development goal |
| SLCP | Short-lived climate pollutants |
| $SO_2$ | Sulphur dioxide |
| TFIAM | Task Force on Integrated Assessment Modelling |
| TSAP | Thematic Strategy on Air Pollution |
| UNECE | United Nations Economic Commission for Europe |

## Glossary of terms used, and their meaning, in this thesis

| 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 contemporary abbreviation of the 1979 UNECE Convention on Long-range Transboundary Air Pollution which was previously 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 BC, OC) |
| Control costs | In this thesis the term describes the costs for reducing air pollution emissions using end-of-pipe technology, altered production technologies, or other similar means. |
| Control option | A specific mean (like EOP technology) available to reduce emissions |
| 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. |
| 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. |

# List of contents

# 1. Introduction

This thesis presents research and reflections on air pollution policy support models used to assist international policy makers in their efforts to reduce problems associated with emissions of air pollutants.[1] To set the stage it is necessary to present the context of the research. This chapter gives an overview of the problem with air pollution in Europe, the policies aimed at curbing the problem, as well as the science developed to support air pollution policy makers. The aim and scope of this thesis is presented at the end of the chapter. Much of the text originates from Åström (2017).

## 1.1. Persistent problems with air pollution

Almost all economic activities cause emissions of pollutants to air. Emissions of the air pollutants covered in this thesis are mainly caused by agricultural activities and combustion of fuels in industrialised societies and are together and separately causing problems with human health, acidification, eutrophication, vegetation damages, and corrosion. Even though air pollution problems currently are more severe on other continents they still cause problems in Europe, despite much progress since the peak pollution years of the 1970's and 1980's.

The World Health Organization (2014a, b) has identified that the largest health risk from environmental causes is driven by human exposure to $PM_{2.5}$ in air. $PM_{2.5}$ in ambient air is mainly constituted of emissions of primary particles as well as of secondary particles (such as ammonium nitrates and ammonium sulphates) formed in the atmosphere and composed from emitted gases such as NMVOC, $NO_x$, $SO_2$, and $NH_3$. Human exposure to $PM_{2.5}$ is associated with premature mortality, heart- and lung related diseases, and many other illnesses (Thurston et al. 2017). In Europe 2012, some 380,000 premature fatalities per year occurred due to $PM_{2.5}$ in ambient air (Lelieveld et al. 2015). By comparison some 25,000 people died in traffic accidents 2017 (European Commission 2018). In Sweden the number of fatalities directly linked to $PM_{2.5}$ exposure is estimated to some 4,700 in 2015 and in addition the number of fatalities linked to vehicle exhaust is estimated to 2,850 (Gustafsson et al. 2018). The latest projections are that air pollution still in 2030 will cause some 194,000 premature fatalities per year in the EU (Amann et al. 2018).

$SO_2$, $NO_x$ and $NH_3$ emissions is also when deposited influencing forest soil and fresh water acidification. Sweden is one of the European countries that still suffer from acidification damages. Although recovery is ongoing, 17 per cent of the Swedish water catchment areas are exposed to acid deposition exceeding critical loads for acidification. The latest Swedish estimates gives that these 17 per cent are expected to decrease to 10 per cent by 2030 (Fölster et al. 2014). Further, reports are now showing biological recovery in European lakes and streams that were previously uninhabitable for several species due to acidification (Garmo et al. 2014). But still several

[1] In the thesis reported here, the term air pollution considers: sulphur dioxide ($SO_2$), nitrogen oxides ($NO_x$), ammonia ($NH_3$), non-methane volatile organic compounds (NMVOC) and fine particulate matter with aerodynamic diameter smaller than 2.5µm ($PM_{2.5}$ or aerosols)

European ecosystems are projected to experience problems with excessive acidification until at least 2030 (Swedish Environmental Protection Agency 2015, Amann et al. 2018, Norwegian Environment Agency 2018).

In addition to effects on human health and acidification 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 waters from emissions of $NH_3$ and $NO_x$ (Sutton et al. 2011), damages from tropospheric ozone to human health, crops, and ecosystems due to emissions of the ozone precursors $NO_x$ and NMVOC (Ebi and McGregor 2008, Van Dingenen et al. 2009), as well as corrosion damages to buildings and materials caused by emissions of $SO_2$ and ozone precursors (Tidblad et al. 2014). In Europe, the evolution of these problems varies. With respect to eutrophication, current trends and projections show declining but remaining problems in large parts of Europe, while the trend for ozone damages is less clear. Results indicate a mixed picture with decreasing peak ozone level concentrations but increasing annual average concentrations. This mixed picture is due to European emission reductions of ozone precursors (which reduce peak concentration) and increased inflow of ozone from other continents in combination with increased methane ($CH_4$) emissions (increasing average concentrations). Trends for corrosion damages show a steady decline over time (Maas and Grennfelt 2016).

## Noticed interactions between air quality and climate change

The characteristics of the climate change and air pollution problems differ in a couple of ways. For example, 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 carbon dioxide ($CO_2$) an atmospheric perturbation time of hundreds of years. The effects also differ in terms of time scales. Some air pollution effects are caused by short term exposure (like acute ozone exposure), and some have an effect that ranges many decades (like long-term exposure to $PM_{2.5}$ and acidification). Climate change impacts act on a longer time scale and through inertia in the global heat circulation system the impacts can last for centuries and more. Linked to this difference in time scales are the geographical ranges of the physical effects. In general, shorter adjustment time scales implies smaller regional effects. Although shared by all populated regions of the world, the physical effects of air pollution are mainly local (cities/countries) and regional (continents) problems, while physical impacts caused by $CO_2$ and $CH_4$ are global. However, both air pollution and greenhouse gas problems are global from social and control technology perspectives.

Despite these differences there are important interactions between air quality and climate change and three major types of physical interaction can be identified from the literature. First, air pollutants and greenhouse gases are often co-emitted from industrial and societal activities including all combustion processes and agriculture. This intertwined feature renders most emission reduction options and instruments to affect multiple gases and there is always some probability of co-beneficial physical effects or trade-offs. Second, air pollutants have when emitted in themselves radiative forcing properties with direct impact on climate change. Third, climate change will, when temperature and precipitation patterns are affected, alter the severity of air quality problems. The following sections presents these interactions in more detail.

### *Co-emission of gases and particles*

As mentioned, emissions of air pollutants and greenhouse gases often stem from the same sources: combustion of fuel and agricultural activities. There are thus direct physical links when considering options to reduce either of the emissions. For an option aimed at reducing impacts of climate change or air pollution, the links can be of mutual benefits (co-beneficial) or antagonistic (causing trade-offs). One typical example of a co-benefit between air pollution and climate change is the use of demand side energy efficiency technology that reduces emissions of both greenhouse gases and air pollutants. Typical examples of trade-off includes fuel shift from fossil to bio fuel which is supposed to decrease $CO_2$ emissions but in the same time risks increasing the emissions of air pollutants (Rafaj et al. 2013, Åström et al. 2013), as well as air pollution control technologies that risk increasing 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 - a fuel shift implemented to reduce $CO_2$ emissions - is another example of a trade-off between climate and air pollution, since diesel cars up until 2017 have been allowed higher $PM_{2.5}$ emissions per kilometre driven than gasoline cars. Furthermore, diesel cars have in real life driving been shown to also have large problems achieving the allowed $NO_x$ emission limits as compared to gasoline vehicles (Weiss et al. 2012), and advanced cheating with engine exhaust systems was exposed in 2015 in the so-called Dieselgate scandal. Cheating and other types of tricks to violate emission limit standards has been calculated to cause some 10,000 premature fatalities in EU28, Norway and Switzerland 2013 (Jonson et al. 2017).

### *Radiative forcing of air pollutants*

Emissions of several air pollutants are identified to have short term and regionally varied impact on climate change. In general, $PM_{2.5}$ (including secondary particles like sulphur aerosols) as well as coarser fractions of particulate matter contributes negatively to the radiative forcing[2] of the climate system, while some sub-fractions of $PM_{2.5}$ like black carbon (BC) as well as tropospheric ozone (affected by emissions of $NO_x$, NMVOC, and $CH_4$) increase the radiative forcing.

On a global scale, todays' atmospheric concentration of particulate matter (including sulphur aerosols) currently counteracts (masks) global warming to an extent corresponding to a radiative forcing (RF) of -0.9 Watt/$m^2$ (current $CO_2$ concentrations has an RF of ~1.82 W/$m^2$). The global average does however hide large regional variation, and the impact of the aerosol components vary (Myhre et al. 2013). As an example, Bond et al. (2013) find that the global average direct RF of

[2] The radiative forcing of an air pollutant or greenhouse gas basically describes its impact on the climate systems' radiative energy balance (solar energy to earth minus heat energy from earth). Radiative forcing is measured in watt per square meter of earth's surface at the tropopause. Forster, P., et al. (2007). Changes in Atmospheric Constituents and in Radiative Forcing. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. S. Solomon, D. Qin, M. Manning et al. Cambridge, United Kingdom, Cambridge University Press.

BC is 0.9 W/m$^2$, with indirect effects adding more unquantified warming. Emissions that act as ozone precursors currently cause an RF of 0.5 W/m$^2$ (Myhre et al. 2013).

The air pollutant gaining most attention recently for its impact on climate change is BC, a soot sub-fraction of $PM_{2.5}$.[3] One tonne BC emissions is considered to have an impact on radiative forcing equivalent to 120-3200 tonnes of $CO_2$ emissions, dependent on climate metric (Myhre et al. 2013), which might be an underestimation (Myhre and Samset 2015). Climate change impact has been identified for all the above presented air pollutants, as well as for the effect of $CH_4$ emissions on ozone formation (Etminan et al. 2016). Collectively, these are therefore often termed short-lived climate pollutants (SLCPs).[4] Control of SLCP emissions have been shown to enable a reduction in the speed of global warming, contingent that $CO_2$ emissions are reduced (Shindell et al. 2012, Bowerman et al. 2013, Shoemaker et al. 2013). However, the generalisation of SLCP impacts is not straightforward. The impacts have a regional nature (Aamaas et al. 2016) and can be located in other regions than the emission source region (Acosta Navarro et al. 2016).

### *Climate change effects on air quality problems*

Climate change is anticipated to bring about warmer temperatures and changes in precipitation patterns. This implies that climate change will influence air quality problems. When focusing on $NO_x$, $NH_3$, and some NMVOCs, European studies find that in general, climate change will induce increased emissions of these pollutants, especially the emission fractions that originate from soils. There is also a risk that climate change will increase vulnerability of ecosystems (Sutton et al. 2015). Another example is that tropospheric ozone concentrations might increase in some regions and $PM_{2.5}$ concentrations might change in many regions, with regional variations (von Schneidemesser and Monks 2013). Research with a Swedish focus have shown that climate change might increase the atmospheric residence time of $SO_2$ and $NO_x$ emissions but decrease it for $NH_3$, with corresponding effect on the importance of long-distance transport of these air pollutants for the Swedish environment. It is expected that recovery of forest ecosystems from excessive acidification is negatively affected by climate change, although other factors (such as biomass harvest) are more important (Munthe et al. 2016).

## 1.2. Policies directed towards the problems

Since emitted air pollutants have a residence time long enough to travel across national borders countries need to cooperate to effectively reduce negative effects of air pollution. In the 1970´s acidification was the first transboundary environmental problem recognized as a serious international and transboundary environmental problem (aggravated partly by the perceived local solution to local problems: tall chimney stacks). This recognition eventually led to the creation of the Convention on Long Range Transboundary Air Pollution (CLRTAP or more commonly Air Convention) in 1979, hosted by the United Nations Economic Commissions for Europe (UNECE).

[3] black carbon is technically only the residual of a certain technique for measuring elemental carbon concentration in air but has become established nomenclature for a specific carbonaceous subfraction of $PM_{2.5}$

[4] Other terms found in the literature are near-term climate forcers (NTCF) and short-lived climate forcers (SLCF).

Today, air pollution is recognised as a global problem and is governed on multiple administrative levels of governance.

Globally, the current framework structure for overall social development, including environmental policy and air pollution policy, is the United Nations 17 sustainable development goals (SDGs), adopted in 2015 as a successor of the millennium development goals. Although not directly mentioned in any of the 17 goals, air quality is directly addressed in three of the 169 sub-level targets and affected by six more (Institute for Global Environmental Strategies 2016). Knowledge is now building up on the interlinkages between the SDGs and air pollution policies driven by the physical interlinkages presented above (Nilsson et al. 2016, Grubler et al. 2018, Rafaj et al. 2018).

In Europe, the 1979 Air Convention and the European Union (EU) thematic strategy on air pollution (TSAP) are the most important policy processes for international agreements on air pollution. The Air Convention has since 1979 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 EU started its efforts to control air pollutants later than the Air Convention, but today it governs the TSAP and several directives that in various ways regulate EU air quality, most recently the 2016 update of the 2001 National Emission Ceilings (NEC) Directive (European Union 2016). The original NEC Directive did set emission targets for 2010 and onwards, whilst the updated NEC sets national emission targets for the EU member states covering the same pollutants as the Gothenburg protocol but with 2030 as a target year.

In Sweden, air quality ambitions are nationally governed under the framework of the Swedish environmental quality objective system, in which three objectives are directly linked to air pollution: Clean Air, Zero Eutrophication, Only Natural Acidification. The most recent development is the 2019 establishment of a Swedish national air quality programme focusing on control of $NH_3$ and $NO_x$ emissions to achieve NEC directive emission levels (Swedish Environmental Protection Agency 2019).

These international and national policies and frameworks and their predecessors have together been a guide to reduce emissions of air pollutants. Aided by structural changes in society, these policy efforts have led to reduced emissions. In western Europe $SO_2$ emissions have by 2010 decreased by 90 per cent since the peak emissions of almost 30 mega tonnes in 1970. $NO_x$ emissions have decreased with 50 percent since peak emissions of around 13 mega tonnes around 1990 (Rafaj et al. 2014a). European emissions of NMVOC and $NH_3$ have decreased with about 50 per cent and 30 per cent respectively since 1990, while $PM_{2.5}$ emissions (for which emission reporting started later) have decreased with about 25 percent from year 2000 levels (Maas and Grennfelt 2016).

## Interactions between air pollution and climate policies

Just as air quality and climate change interact on a physical level, so does the policies implemented to reduce the problems. In this case however, the interactions rather relate to economics and policy target achievements. In general, the implementation of climate policy can reduce the subsequent costs of implementing air pollution policy, but an integrated approach can reduce costs even more. However, the size of the cost reduction is sensitive to the instrument chosen to implement climate policy. If emission reductions rather than cost savings are prioritised it is possible to achieve beneficial policy interactions between climate and air pollution policies. But as is the case for control cost interactions, the size of the potential benefits of policy interaction is sensitive to the instruments used in the respective policies. Further, the policy community has on several occasions misinterpreted the opportunity for co-beneficial policy effects and proposed climate-policy-only approaches. The following section presents these policy-related interactions in more detail.

### *Emission control cost interactions*

Policies used to control air pollution and climate change implies co-benefits or trade-offs on costs for the economy (Apsimon et al. 2009). In general, climate policies are found to be co-beneficial for air pollution control costs, but the size of the co-benefit is largely dependent on the climate policy strategy chosen and how ambitious its GHG targets are.

Earlier studies showed that an expected implementation of the Kyoto protocol (the first global quantified climate agreement) in the EU could enable economic co-benefits between air pollution control and greenhouse gas (GHG) emission control by 2010. The size of the economic co-benefit was however dependent on the degree of integration between climate and air pollution policy. When analysed as policies implemented sequentially, the reduced costs for air pollution control of a Kyoto protocol climate policy would amount to 10-20 percent of total GHG control costs (Syri 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 et al. 2006). In addition to being determined by the level of integration and the ambition of climate policies, also the policy instrument 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 and air pollution control (Syri et al. 2001, van Vuuren et al. 2006, Rypdal et al. 2007). Newer studies, analysing economic co-benefits in 2030, also find economic co-benefits of integrating air pollution and climate change policies (McCollum et al. 2013, Rafaj et al. 2013).

### *Interactions of policies*

By considering the potential for reducing air pollution emissions instead of lowering control costs one can study co-benefits and trade-offs on policy targets between air pollution and climate policies. Such studies show that climate policies often lead to co-benefits on environmental and human health effects due to reduced emissions of air pollutants. For example, if the EU were to strive for a two-degree climate policy target, this would reduce human health effects by some 70 percent compared to a no-climate scenario by 2050 (Schucht et al. 2015), or by some 35 percent compared to a Kyoto protocol baseline scenario (Rafaj et al. 2013). Other studies have shown that these types of co-benefits could continue to increase until at least 2100 (West et al. 2013). However, as was the case for economic co-benefits, the climate policy mechanism will affect the size of the emission co-benefits. Air pollution effects are unevenly distributed geographically, and GHG emissions trading might reduce the co-benefits in Europe. Further, climate policy alone is not considered enough to achieve air pollution policy targets (van Harmelen et al. 2002, Rafaj et al. 2013, Amann et al. 2014b). There is however today no international policy that takes a fully integrated approach by setting emission targets for both air pollutants and greenhouse gases.

In fact, the opportunity for policy co-benefits, which has been communicated by scientists for years, might potentially have had the unintended effect of providing an argument for focusing the policy process entirely on GHG control. Such an argument was raised in the negotiations preceding the NEC directive in the early 2000 (which coincided with the Kyoto Protocol implementation negotiations), and during the first (and cancelled) effort to review the NEC directive in 2005-2007 (which coincided with the EU negotiations for the 2020 Climate & Energy package[5]). The same argument was used again in December 2014 when the European Commission (EC) suggested modifying (retracting) the proposal for an updated 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."*[6] (European Commission 2014a).

In other words, the EC/EU has in three cases considered air pollution policy development as superfluous and used policy efforts made to control GHG emissions as rationale. This even though analysis gives no support for such cancellation. Most recently, (Amann et al. 2014b) explicitly showed that the 2030 EU Energy and Climate Package would decrease 2030 emissions of air pollutants with only 4-10 per cent compared to a 2030 baseline.

## 1.3. Scientific support to policy makers

The expected effect of an air pollution policy proposal is by no means intuitive or calculable on the back of an envelope (nor a climate policy proposal for that matter). Therefore, the European transboundary air pollution policy process has since the 1970's been relying on scientists to develop knowledge and on computer models to assess effects of policy proposals. The information flow between scientists and policy makers has been going both ways, and the European air pollution science and policy development is a typical example of what has been dubbed co-production of knowledge between science and policy (Tuinstra et al. 1999, Tuinstra et al. 2006, Tuinstra 2007, Dilling and Lemos 2011, Reis et al. 2012).

Science have helped shape the formulation of the protocols under the Air Convention and the most recent, the Gothenburg protocol, has an effect-based[7] focus where future effects on the environment and human health as well as cost-effective emission control strategies are identified using models. The EU efforts to reduce negative effects of air pollution have developed on a similar path as the Air Convention, although often focusing on specific sectors or fuels. The newer

[5] -20% GHG compared to 1990 by 2020, 20% renewable energy, 20% improvement of energy efficiency compared to a 2007 baseline projection

[6] -40% GHG compared to 1990 by 2030, 27% renewable energy, 30% improvement of energy efficiency compared to a 2007 baseline projection

[7] Effect-based: policy objectives are set on environmental and health effects instead of on emission levels

directives, such as the 2001 NEC Directive and the 2008 Air Quality Directive, are effect-based and influenced by modelling of environmental and economic effects of policy proposals.

### *Indicators*

Almost since the beginning of international air pollution 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 in the assessments 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 et al. 1995) the impact assessments can model potential effects on excessive acidification and eutrophication of ecosystems from reduced emissions of air pollutants. Through the indicators phytotoxic ozone dose (POD) (Emberson et al. 2000), and the accumulated amount of ozone over the threshold value of 40 parts per billion (AOT40), the effect on ozone damages on vegetation can be assessed. Through progress in materials science standardised links between corrosion damages and air pollution emission levels can be estimated (Tidblad et al. 2014). By the late 1990´s and early 2000´s the epidemiological knowledge-base was advanced enough (Pope et al. 1995, Pope et al. 2002) to allow for modelling of human health effects of air pollution and changing air pollution concentrations. All these indicators are enabled by regular monitoring and modelling of air quality (Simpson et al. 2012, MSC-West et al. 2017), experiments and modelling of health and ecosystem effects from air pollution (Lundbäck et al. 2009, CCE 2016), as well as research coordination efforts mainly within the Air Convention (Reis et al. 2012).

### *Integrated assessment models to account for system-wide effects*

Given the multiple facets of air quality problems, integrated analyses are used to ensure that any policy impact assessment reasonably estimates the multiple effects, geographical differences, and varying socio-economic development of relevance for air pollution policy. Further, scenario analysis is used since structural changes in the economy, changes in fuel use, and changes in industrial production all affect future emission levels. To meet these demands, integrated assessment models (IAMs) such as UKIAM (Oxley et al. 2003), MERLIN (Reis et al. 2003, UCL 2004), RAINS (Amann et al. 2004), and GAINS (Amann et al. 2011b, Kiesewetter et al. 2014, 2015a) have been developed.[8] In this thesis, the term AP-IAM is used to separate this group of models from the currently more well-known IAM models used to analyse climate change policies. The AP-IAM models build upon the knowledge produced mainly in the above-mentioned research fields. Basically, an AP-IAM is used to specify which control options that should be implemented to control emissions from European countries and how large the control costs would be for a given policy target. Commonly the options available for consideration are different end-of-pipe options. Current AP-IAMs can consider: that several pollutants contribute to one or several environmental problems and climate change; that ecosystem sensitivities vary between countries; that emission dispersion and mixing in the atmosphere follow certain meteorological conditions, and that the

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[8] UKIAM: United Kingdom Integrated Assessment Model;
MERLIN: Multi-pollutant, Multi-Effect Assessment of European Air Pollution Control Strategies: an Integrated Approach;
RAINS: Regional Air Pollution Information and Simulation;
GAINS: Greenhouse Gas - Air Pollution Interactions and Synergies

economic structure of a country affects the ability and cost of emission reduction. Through the focus on socio-economic costs, the models can also identify in which sectors options should be implemented to provide lowest costs for the entire economy. Correspondingly, most European air pollution policy impact assessments are done with AP-IAMs. Examples are the Air Convention Gothenburg Protocol (CLRTAP 1999, Amann et al. 2011b, c), and the EC proposal for a clean air policy package (CAPP) (European Commission 2013a).

### *Cost-benefit analysis to check soundness of policy proposals*

The discipline of environmental economics has grown in importance since the 1970's and onwards, and with that numerous monetary valuations of non-market effects of human activities has become available. Correspondingly evaluation of policy proposals via appraisal and comparison of costs and benefits (Cost-Benefit Analysis, CBA) has been made possible. This in turn has enabled socio-economic impact assessment (in practice CBA) to become mandated by law in many countries. For the European Union, the requirement for official bodies to use CBA is explicitly mentioned in EU Regulation No 1293/2013 and 1303/2013 (European Union 2013). In Sweden, assessment of socio-economic outcomes prior to public sector policies or investments is regulated in SFS 2007:1244 (Kriström and Bonta Bergman 2014). Guidebooks are continuously developed to ensure consistency and that latest knowledge is considered (Swedish Environmental Protection Agency 2003, European Commission 2014b, Kriström and Bonta Bergman 2014, Swedish Road Administration 2015, 2018).

In an air pollution context, the policy support material to the Gothenburg protocol as well as the EU Clean Air for Europe (CAFE) programme both used air pollution CBA (AP-CBA) as complementary analysis to verify that proposed ambition levels could be justified from a socio-economic perspective (Holland et al. 1999, Holland et al. 2005). For the EC, AP-CBA are often made through combining GAINS model results with results from the benefit assessment tool Alpha Risk-Poll (ARP) (Holland et al. 2013, Schucht et al. 2015). CBA has also been used to evaluate more specific policy initiatives, such as the EU fuel quality Directive (Bosch et al. 2009) and local air quality policies (Miranda et al. 2016).

### *The latest addition, using models to propose policy targets*

For the latest addition to European air pollution policy, the 2016 update of the NEC Directive (European Union 2016), the scientific support shifted approach. Earlier support had used a cost-effective (minimizing control costs) and effect-based approach, but now the support used a cost-efficiency (maximising social welfare) and effect-based approach, thus increasing the precision of the results from the models used. With this shift the European Commission, who proposed the update (European Commission 2013b), chose to use AP-CBA to obtain initial values for the emission ambition level in their policy proposal. In other words, in 2013 the EC shifted from using models for appraisal of costs and benefits of achieving a proposed target into using models for identification of the desirable target to be proposed. They also allowed the models to be used for a very precise prescription of a policy proposal (2030 emission levels of five pollutants in 28 countries). Figure 1 show the modelled EU marginal costs and benefits following increased ambition from a 2025 emission levels in a current air pollution legislation (CLE) scenario and a scenario in which all available control technologies are used (MTFR). The advice from the model

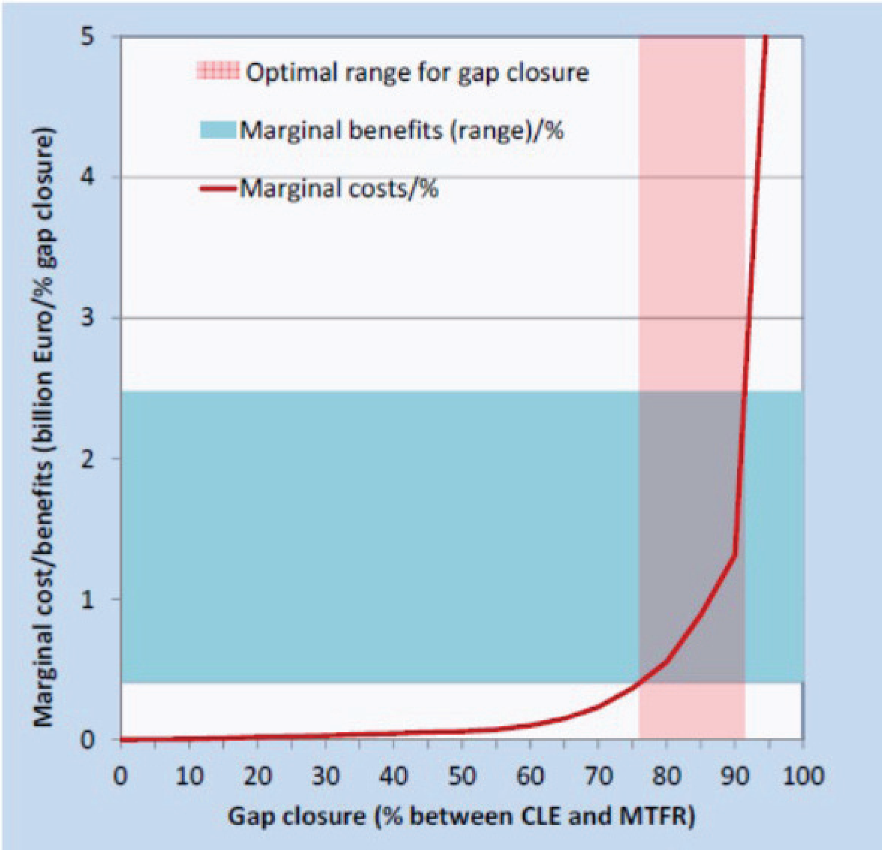


**Figure 1: Marginal emission control costs and marginal health benefits in 2025 as estimated in the EC policy support material. Copied from Amann et al. (2014a)**

results to the EC is seen in the intersection of the marginal control cost (MC) and marginal benefit (MB) curves (the red shadowed area, corresponding to a model ambition level of 76-92 per cent closure of the gap). Within this area, the modelled emission levels are considered cost-efficient.

This new approach to air pollution policy targets represents an expanded use of available knowledge, and a recognition of a knowledge-based approach to environmental policy. It also increases the extent to which European air quality is dependent on the concept of AP-CBA and high precision computer simulation models.

## 1.4. Aim and scope of this thesis

My interest in the research supporting the EU air pollution policy process has been growing since 2006. At that time, it was recognised by Swedish officials that the GAINS model is so influential for European air pollution policy that domestic expertise in the model would be beneficial for Sweden. A research consortium led by IVL Swedish Environmental Research Institute won a bid for a new research programme, the Swedish Clean Air Research Programme, within which Swedish GAINS modelling activities were initiated in 2006. It was under the auspices of this research programme that I with my background in environmental science and environmental economics started learning the GAINS model and air pollution policy support modelling.

An important venue for updates on state-of-the-art European AP-IAM modelling is the UNECE Task Force on Integrated Assessment Modelling (TFIAM), a forum to evaluate methods and tools for integrated assessment of air pollution emission control. After having regularly attending these meetings for a couple of years I was in October 2013 appointed by Sweden to share the chairmanship of the TFIAM with the Dutch co-chair, a role I still hold. The role as co-chair of TFIAM has given me the opportunity to observe and participate in the development of European air pollution policy support models, as well as noting the way in which results from these models are received by policy makers and other stakeholders. Both my research background and policy background have helped frame the direction of the research presented in this thesis.

The overall aim of the research in this thesis is to add to the scientific basis of the air pollution policy support models used in Europe. Given that the models are now used to give precise and prescriptive results, increased scrutiny of the models is called for. More specifically, the common theme of the research is that it checks the robustness of policy recommendations from AP-IAM and AP-CBA. Paper I study whether the relative cost-effectiveness of SLCP control options (i.e. ranking) is affected by the choice of climate metric used when calculating cost-effectiveness. Paper II study how much of the decoupling of $SO_2$ emissions from economic growth that was due to dedicated $SO_2$ control options. Paper III study to what extent differences in economic perspectives affect the modelled costs of reducing emissions, where emission reductions would be recommended, and which pollutants that should be in focus. Paper IV study the costs and benefits in 2030 of implementing a nitrogen emission control area by 2021 in the Baltic and North seas as a mean to reduce adverse environmental and health impacts from $NO_x$ emissions. A synthesis of paper III and IV then allows for a comparison of the cost-effectiveness of land-based emission reductions and emission reductions from international shipping, and thereby a robustness check of the choice to only include land-based emission control technologies in policy analysis. In addition to papers I-IV, which focus on methods, Paper V discusses and reflect on the methodology utilised in policy support modelling, and its correspondence with policy making.

This thesis also utilises the opportunity to document experience-based reflections on methods and results, as well as wider reflections on the research's relevance to society and future research. Hopefully this documentation can be useful for future air pollution policy and policy support research.

The research is of integrated nature but always at least partly directed towards economics and integrated assessment modelling. It covers the air pollutants mentioned above and in some cases greenhouse gas emissions. The research is both retrospective and prospective in nature. Further, to ensure relevance to the policy process, the methods used are the same as in current EU air pollution policy support science, but more limited in regional coverage. These methods are well established and regularly peer reviewed. The most extensive peer review as of lately was done in 2004-2005 (Grennfelt et al. 2004, Krupnick et al. 2005). Since then a smaller internet consultation have taken place in 2008 and a review of the epidemiological evidence of health effects from air pollution was done in 2013 (WHO 2013a, b).

# 2. An overview of the methods used

Decomposition analysis, integrated assessment models, as well as cost-benefit analysis are all methods used to support UNECE and EU air pollution policy makers. In an air pollution policy context, the methods are usually all utilising physical data or scenarios on societal activities required for industrial production and household services, whereas only the latter two always utilise economic data as well. The methods also differ with respect to which type of questions they answer. Decomposition analysis is used to single out which specific type of activity that is most responsible for emission changes, integrated assessment models are used to analyse cost-effective strategies to reduce adverse environmental effects, whilst cost-benefit analysis is used to either check socio-economic soundness of existing initiatives, or to find cost-efficient (welfare-maximising / optimal) emission levels. The following text provides more details, some of which originates from Åström (2017). For complete method descriptions see the papers I-IV.

## 2.1. Decomposition analysis

Paper II utilise an extended version of decomposition analysis to identify to what extent specific Swedish $SO_2$ policies and instruments contributed to the reduction of $SO_2$ emission levels in Sweden 1990-2012. Decomposition analysis of emission pathways clarifies the relative importance of the driving forces, such as economic growth, structural changes, fuel shifts, other policies etc., behind the emission pathways and their development over time (Hoekstra and van der Bergh 2003). 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 pathway. Following this, all drivers but one are kept at the base year values and an alternative pathway is calculated. The impact of the driver kept constant is then identified through subtraction of emissions in the alternative pathway from emissions in the baseline pathway. Retrospective analysis of historical data is the most common setting for decomposition analysis, but there are examples of decomposition analysis done prospectively (Rafaj et al. 2014b). There are different types of decomposition analysis, and in the literature, it is common to separate between structural decomposition analysis, index decomposition analysis, and the mix of the two: Divisa index. For a more comprehensive description of these, see Hoekstra and van der Bergh (2003).

The extended decomposition analysis utilised in Paper II is based on Rafaj et al. (2014b) and use detailed Swedish energy, industry, and $SO_2$ emission statistics for 1990-2012 to analyse the relative impacts on $SO_2$ decoupling from economic growth due to 1) structural changes in the overall economy, 2) fuel use changes (changes in total fuel demand and fuel mixes), 3) changes in industrial productivity, and 4) emission factor changes. In addition to earlier decomposition analyses, Paper II also links the results from the decomposition analysis to actual $SO_2$ policy instruments implemented in Sweden for the period. This link is enabled by comparing timelines of legislation development and timelines of changes in $SO_2$ emission factors (unit $SO_2$ per unit activity) for the affected sectors or fuels while controlling for confounding factors.

A retrospective decomposition analysis is per definition a counterfactual analysis, a concept that deserves some attention. As one can envision from the name, a counterfactual analysis asks the question "What if what really happened didn't happen?". However, 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 (Ferraro 2009). This is partly due to the national scale of the policies (which omits the use of control groups) but also due to omitted consideration of evaluation needs 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 suitable tool for environmental policy evaluation to ensure that potential effects of other confounding factors are considered when analysing the effect of a policy intervention (Ferraro 2009).

## 2.2. Integrated assessment of air pollution policy

Paper III studies the robustness of integrated assessment models through a sensitivity analysis of two of the parameters in the economic analysis, the interest rate on investments and the lifetime of investments. To better enable the reader to follow the discussion related to the economic aspects of integrated assessment modelling, the concept of cost effectiveness analysis first needs to be presented.

### *Cost effectiveness analysis*

Cost effectiveness analysis (CEA) is, within the scope of this thesis, used to identify which control options to use so that a desired target can be met at lowest control cost. Dependent on model approach, cost effectiveness can be analysed with different monetary metrics. In the context of AP-IAM, control costs of an option 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. CEA is used in Paper I and Paper III.

### *Integrated assessment modelling*

The AP-IAM discussed in this thesis is the GAINS model (Amann et al. 2011a, Kiesewetter et al. 2015b), developed by the International Institute for Applied Systems Analysis (IIASA). The GAINS model is an update of the older RAINS model and is developed in different versions. The versions discussed in this thesis are the European and Fenno-Scandinavian versions focusing on control of air pollutants.

The GAINS model is a bottom up AP-IAM developed to analyse how future air pollution emissions can be reduced to achieve specified positive effects on the environment and human health to the lowest cost. The model is constructed of the main components: exogenous scenario data on polluting activities; database information on emission factors, control options, and emission control costs; linear form calculations of emission dispersion and deposition over

Europe; and exogenous data on ecosystem sensitivities and on population demographics. These components then enable calculation of scenario-specific results on emissions, emission control costs, as well as environmental and human health effects.

Several separate research disciplines and models feed in to the GAINS model (Figure 2). Exogenous scenario data on polluting activities is taken either from European scale energy system models and agricultural models such as POLES, CAPRI, and PRIMES (Russ et al. 2009, Britz and Witzke 2014, NTUA 2014), or from national scenarios supplied by national experts. The linear form calculations of emission dispersion are based on calculations with the chemical transport model EMEP (Simpson et al. 2012) and the exogenous data on ecosystem sensitivities was up until 2017 provided by the then discontinued Co-ordination Centre for Effects (CCE) of the Air Convention (Posch et al. 2012, Hettelingh et al. 2017).

To achieve a result with specified effect on human health and the environment to the lowest cost the GAINS model minimizes costs 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 constraints on how fast a technology can be phased in or out). 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 et al. 2013).

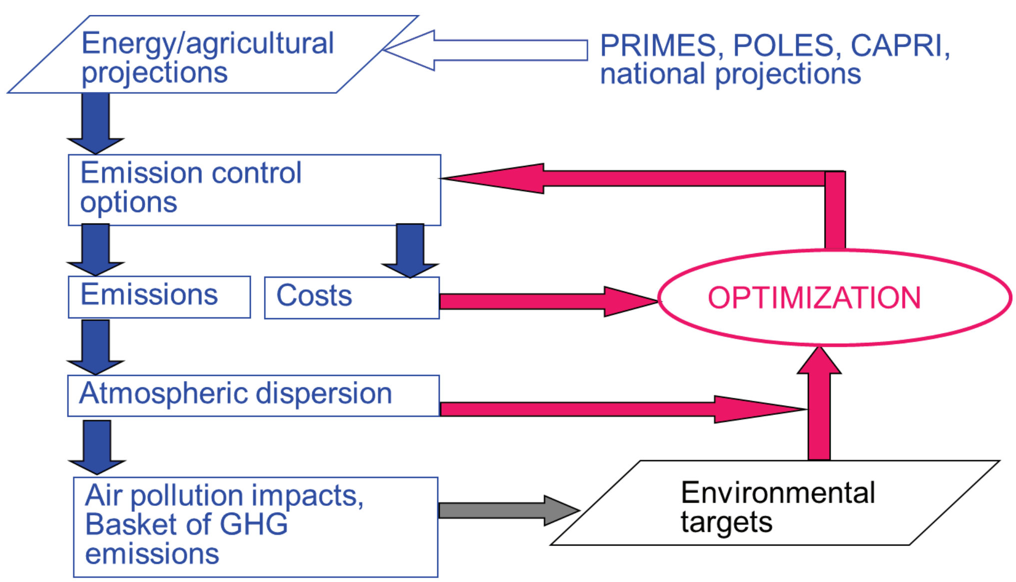


**Figure 2: The data and information flow chart for the control cost optimization of the GAINS model. Copied from Amann et al. (2004)**

The GAINS model utilises a rich description of control options when minimizing control costs. Compared to for example some economic equilibrium models, one can consider the GAINS model to use a techno-economic approach to costs of emission reductions, which ensures physical consistency of modelling results. A scenario version of the GAINS Europe model is freely available online.[9]

## 2.3. Air pollution cost-benefit analysis

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 and many guidelines have been written on how to do a CBA. In a typical manual, a CBA should include the following steps (adapted from Boardman et al. (2001)):

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

A case-specific monetization of the environmental 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 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 (Desvousges et al. 1998). The transfer of benefit values can be done through different levels 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 economic parameters such as GDP per capita and purchase power parity, 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 et al. 2015).

There are two main versions of CBA applied for policy analysis, one version based on optimization and one version based on scenario comparisons. As was mentioned above, AP-IAM applies cost-effectiveness analysis, whilst AP-CBA in the optimization version rather should be

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[9] http://gains.iiasa.ac.at/gains/EUN/index.login?logout=1&switch_version=v0, access is subject to registration

considered an application of cost-efficiency analysis. Cost-effective and cost-efficient should not be considered two words for the same concept. A cost-effective choice is the choice with lowest cost to reach a given target. In contrast, a cost-efficient choice is a choice that gives human health and environmental effects so that net socio-economic benefit of emission reduction is maximised (adapted from OECD (2011)). In AP-CBA it is presumed that the demand for environmental quality and human health is dependent on the cost of satisfying the demand and most often it is assumed that each incremental improvement is worth less than the previous. It is also assumed that emission control costs increase with increasing policy ambition. 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 (optimal) for society. AP-CBA thereby identifies cost-efficiency as in contrast to cost-effectiveness identified with CEA and AP-IAM. In this thesis the terms 'cost-effective strategy' and 'cost-efficient solution' are used to help separate the concepts.

The second main version of AP-CBA is to identify which of the available options (or policies) that would give highest available net socio-economic benefits for society. The results from such an AP-CBA often show the ratio of total benefit over costs (B/C ratio). If the B/C ratio is above one, the solution gives net socio-economic benefits, although it might not be optimal. This latter version can be considered useful if many options are available to reach the same target or if the control options studied are non-additive. This latter version of AP-CBA is the one used in Paper IV.

# 3. Results and reflections on the methods used

The research shows a mixed picture with respect to robustness of air pollution policy support methods. There are also a couple of limitations with the methods themselves that constrains the interpretation of results from air pollution policy support models. This chapter does not reiterate results and conclusions drawn in the individual papers (for that the papers are recommended), but rather interpret the results in the context of the specific aims of the research presented in this thesis.

## 3.1. Synthesis of results from Paper I-IV

The results from the papers included in this thesis have direct bearing to the main research theme of concern: robustness check of models and concepts used in current research supporting air pollution policy. These results are grouped in pros and cons in the following section.

### *The current AP-IAM concept appears robust with respect to the use of climate metrics and model focus on end-of-pipe options to reduce $SO_2$ emissions.*

To start with, Paper I present a parameter sensitivity analysis over climate metrics for nine SLCP options available in Sweden. The result shows that the recommendations from AP-IAM models on which emission control options to implement for cost-effective SLCP emission reduction are robust with respect to the climate metric used when converting the SLCP reductions into $CO_2$ equivalents. Figure 3 show the distribution of results when nine SLCP control options are ranked in order of climate cost-effectiveness, while the climate metric used to calculate climate impact is varied. The calculations supporting Figure 3 consider six pollutants with three possible values of climate impact for each of the eight climate metrics. Correspondingly for each metric all options have 729 ($3^6$) possible cost-effectiveness values, and corresponding number of relative rank outcomes. The relative cost-effectiveness of the options (i.e. ranking) is stable over the metrics. The only caution regards measures that affect $NO_x$, which has uncertain but varying short- and long-term climate impact, which in turn makes the $NO_x$ values for one climate metric shift sign from negative to positive. This change in sign can be seen to destabilise the cost-effective ranking of measures affecting $NO_x$.

Another result that supports the robustness of the GAINS model comes from Paper II which studies past policy instrument effect on $SO_2$ emission reductions in Sweden. The results show *inter alia* that changes in emission factors (mainly due to dedicated end-of-pipe emission control) were responsible for at least 48 per cent of national $SO_2$ emission reductions in Sweden 1990-2012 (Figure 4). This despite that already by 1990, Sweden had reduced $SO_2$ emissions from a peak of 930 ktonne in 1970 (Broström et al. 1994) to 105 in 1990. Sweden is an international front-runner in $SO_2$ emission reduction, between 1970 and 2012 Sweden reduced $SO_2$ emissions by 97 per cent, compared to 90 per cent for western Europe and 70 per cent for Eastern Europe until 2010 (Rafaj et al. 2014a). It is therefore likely that end-of-pipe still can contribute with much of future $SO_2$ emission reductions in other European countries, and thereby that the GAINS model focus on end-of-pipe options can be at least partly supported for $SO_2$.

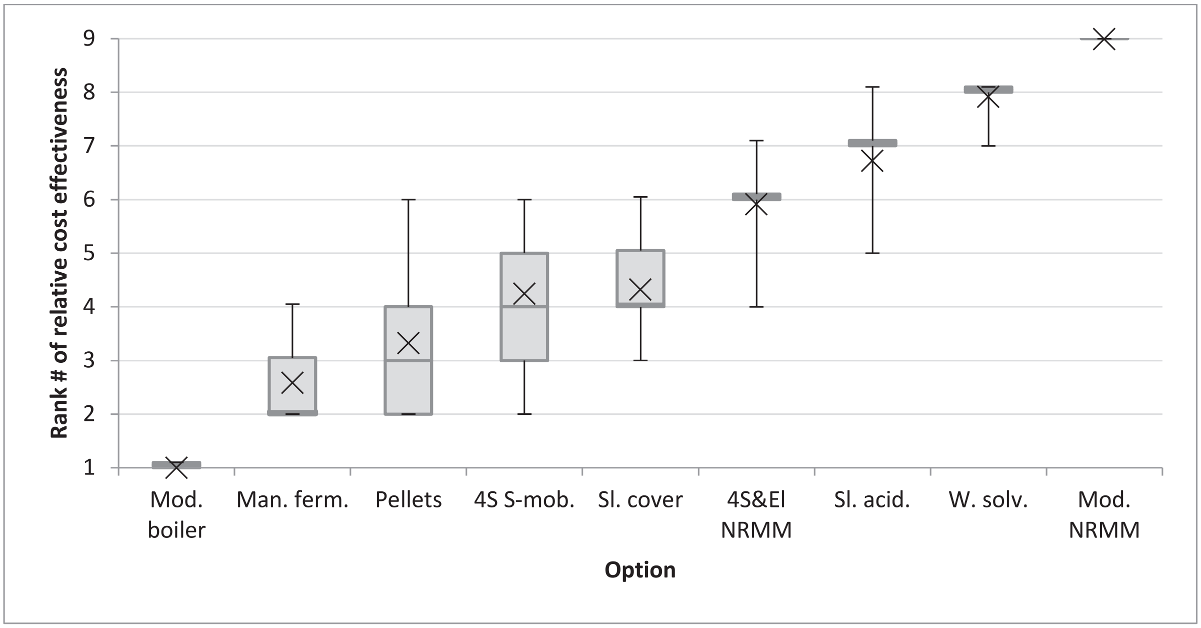


**Figure 3: Distribution of relative ranking (#1 to #9 on the y-axis) of cost effectiveness (€$_{2010}$ /tonne $CO_{2eq}$) for each of the nine options analysed in Paper I (on the x-axis). The grey box shows the range of ranks for the 2$^{nd}$ & 3$^{rd}$ quartile of the results, the grey line in the boxes shows the median rank, the cross shows the average, and the error bar shows the 90$^{th}$ percentile range.**

Together, Paper I and II suggest that the GAINS model is robust with respect to cost-effective ranking of SLCP control options, and that it is reasonably robust with respect to only considering end-of-pipe emission control options for $SO_2$.

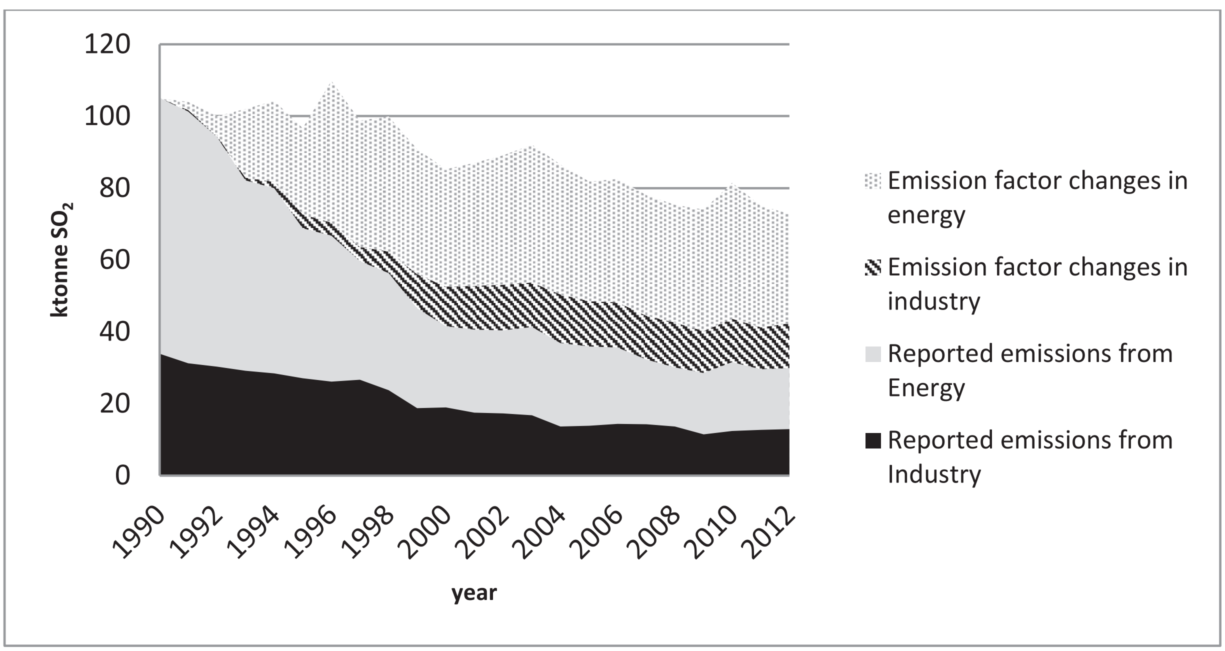


**Figure 4: Swedish reported $SO_2$ emissions as well as counterfactual emissions if emission factors would have remained constant on 1990 levels. The two bottom areas show the emission trend from industrial processes (Industry) and from fuel combustion in energy and transport (Energy). The two top areas show the counterfactual emission trajectories for Industry and Energy sector emissions if emissions factors would have remained constant at 1990 values over the period**

***The current AP-CBA concept doesn't appear robust with respect to investment perspectives and omission of international shipping options.***

The situation is however not as reassuring when it comes to the robustness of model outcome with respect to investment perspectives and omitted sources of pollution control. In Paper III a Fenno-Scandinavian version of the GAINS model is used to analyse if different investment perspectives would affect the cost-effective ranking of emission control options for available policy ambition levels. Two investment perspectives are analysed, a social planner perspective and a corporate perspective. These differ in the assumed interest rate on investments (4 per cent for the social planner and 10 per cent for the corporate) and economic lifetime of investments (economic lifetime = technical lifetime for the social planner and <10 years for the corporate). Although the results vary over policy ambition level, there are several ambition levels for which the choice of investment perspective determine the cost-effective ranking and modelled use of emission control options, as indicated by differences in costs to reach a specified ambition level (Figure 5). In other words, the GAINS model results are in this scenario analysis not always robust with respect to interest rates and payback time of investments and there is reason to argue for sensitivity analysis over these cost parameters in future policy analysis.

Another example of when the robustness of the AP-IAM and current AP-CBA model results can be questioned comes from Paper IV in which the 2030 social costs and benefits from a 2021 implementation of a nitrogen emission control area (NECA) in the Baltic and North seas is analysed.

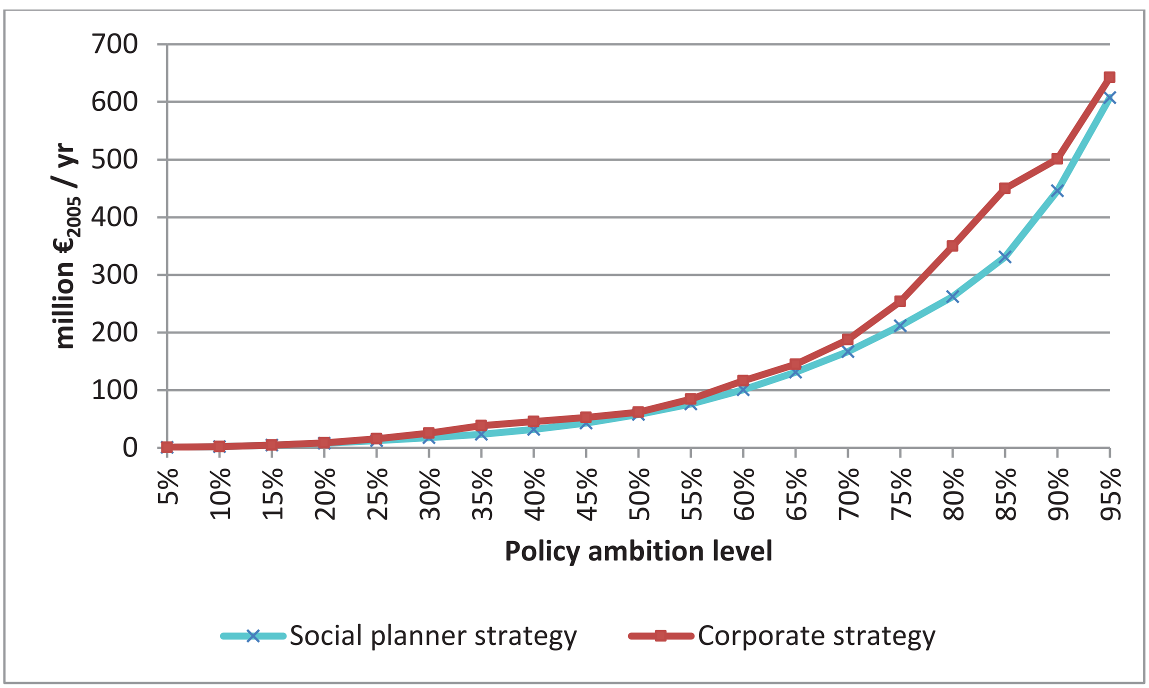


**Figure 5: The Fenno-Scandinavian social planner emission control cost curves in 2030 when cost optimization is based on either a social planner perspective (Social planner strategy) or a corporate perspective (corporate perspective). The maximum surplus cost of letting the corporate strategy determine technology use is found at an 85% ambition level, where they correspond to 120 million € per year**

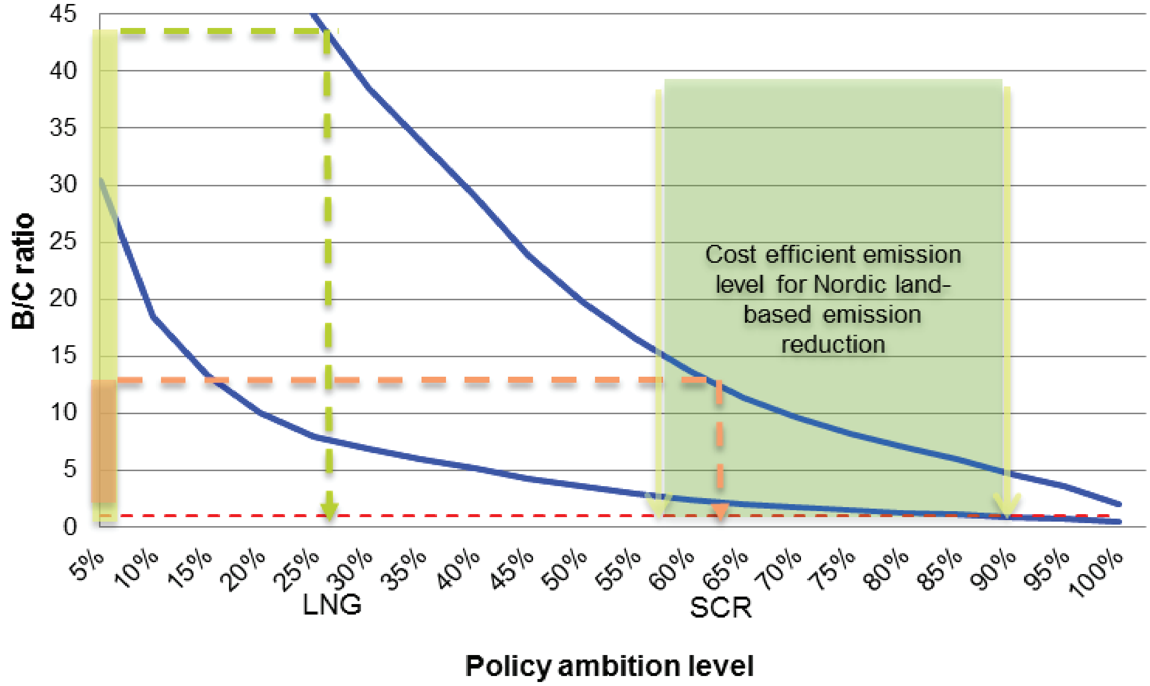


**Figure 6: Benefit/cost ratios of reducing Fenno-Scandinavian emissions of air pollutants in 2030 from land based sources (two blue lines represent low and high range), or through reduction of emissions from international shipping in the Baltic and North seas via LNG propulsion engines (green dotted lines) or via selective catalytic reduction (SCR, brown dotted lines). The low-high ranges of B/C ratios for the shipping options are represented with the coloured bars on the y-axis (green for LNG and brown for SCR). The cost-efficient solution for land-based emission reduction is represented with the green shaded area.**

A re-analysis of the Paper IV B/C ratios and comparison with corresponding B/C ratios for the emission reductions analysed in Paper III, show that the use of Liquid natural gas (LNG) propulsion technology to reduce air pollution problems can be more beneficial for the Northern European societies than land-based Fenno-Scandinavian emission reductions already at a policy ambition level of 25 per cent implementation of available land-based technologies (Figure 6). Other technologies, such as selective catalytic reduction (SCR) would be preferable to land-based options from a 65 per cent Fenno-Scandinavian ambition level and onwards.

One extension of this research is that the results presented in Paper IV are solid enough to serve as input to an extension of the GAINS Fenno-Scandinavian optimization so that this now includes shipping options in addition to options from land-based sources, research that is currently ongoing. Further, the data produced in Paper IV has already been used in new policy research on implementing sulphur and nitrogen emission control areas in the Mediterranean Sea (Cofala et al. 2018).

## 3.2. Critical review of the methods

The methods used in air pollution policy support models require attention since these affect which conclusions that can be drawn from the analyses. The nature of the limitations in the methods differ slightly. In general, decomposition analysis is limited by the independence of emission drivers, whilst AP-IAM is limited by data availability and economic rigidity, AP-CBA on the other hand is limited by data availability and in the dealing with temporal issues.

### *Limitations of decomposition analysis*

In the literature – and in Paper II – there is little attention to the presumption that the driving forces develop independently over time and that one driving force could develop whilst the other remain constant (the all-else-equal, *ceteris paribus*, condition). A typical quote from Stern (2002) serves as example of this approach.

> *"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."*

One problem with this approach is that economic growth is presented as independent of structural/technological changes in the economy, whereas in economic literature economic growth is considered to be driven much by structural/technological changes (Solow 1956, Romer 1990, Schumpeter 2003 (1942), The Royal Swedish Academy of Sciences 2018). Further, in historical data the changes that occurred are known, so to ignore them by assuming the temporal *ceteris paribus* condition in the analysis of historical data can reduce the validity of the results from a decomposition analysis.

The method is also sensitive to the assumed order of the emission driving forces. In Paper II, and other similar analyses, it is assumed that economic growth is the primary driver of emissions, structural change the secondary, fuel efficiency the third, fuel shifts the fourth, and emission factors the fifth driver. As is shown in the Paper II sensitivity analysis, the drivers' effect on emissions can be sensitive to their relative order, but this is an aspect not usually considered in these analyses.

Finally, and of special relevance for Paper II. Given that decomposition analysis doesn't explain causality between emission drivers and $SO_2$ policy instruments, and that there is disagreement in the literature on the physical functionality of $SO_2$ policy instruments, a limiting assumption is necessary. Paper II satisfy with the assumption 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 effect on other emission driving forces as well. Therefore, the reported impact of $SO_2$ policy instruments on $SO_2$ emissions in Paper II should be considered as underestimations.

### *Some commonly noted limitations of CEA /AP- IAM*

There are a couple of limitations of AP-IAM models. The limitations brought up in this thesis adhere to the economic dimension of the GAINS model, most of which should be fully or partly applicable to other AP-IAM models as well.

Most prominently there are several emission control measures not yet included in the optimisation. There are many climate change control measures that can imply co-benefits with air pollution control such as fuel shifts, energy efficiency improvements, and demand side management. There are also measures that due to limited understanding, data availability, and comparability are excluded from the optimization. This group contains structural changes (transport planning, urban development plans, etc.) as well as behavioural changes.

Another perspective on the above-mentioned problem with missing options is the fact that techno-economic models such as GAINS tend to miss out on adaptations made by economic agents in response to price signals. In some other models adaptation to price signals is commonly captured via the use of price elasticities, as in the EMEC model (Berg et al. 2012). The use of elasticities allows for accommodation of observed market behaviour, but has problems identifying the casual physical mechanisms leading to emission reductions. In other words, using elasticities to estimate emission reduction potentials might lead to larger emission reduction potential in the models, but with reduced technical and physical explanatory capacity as a trade-off. Technology learning and its impact on control costs can also be considered as economic adaptations not considered in the European air pollution version of GAINS.

Further, the GAINS model database used in the cost optimization currently contains control options from land-based sources. Currently only the Fenno-Scandinavian version of the GAINS model allows for cost optimization that considers options both at land and at sea.

### *Commonly discussed limitations of CBA*

The critique and discussions surrounding CBA spans over many dimensions and covers many aspects. For a thorough philosophical overview of the critique see Frank (2000) and for recent overviews see Hwang (2016) and Åström (2017). Since the discussions do not focus on any specific environmental problem, the general term CBA is used in most of this chapter. When reviewing the literature one can identify three broad themes of critique: measurement problems, discounting issues, and ethical (including distributional) concerns.

The act of measuring the prices of non-market goods and services has proven to be difficult. In principal there are two main methods, observations (revealed preference methods) or interview experiments on hypothetical markets (stated preference methods). Ensuring that revealed preferences or stated preferences actually include the entire good or service in focus but not more can require problematic assumptions that renders interpretation to be difficult (Frank 2000), and derived values can be context dependent (Horowitz and McConnell 2002, Tunçel and Hammitt 2014). Furthermore, it is recognised that several goods and services of concern in a CBA are often not measured. Some emphasise that this can be because the derivation of a single-unit metric when comparing different types of goods and services just is not feasible: the incommensurability problem[10] (Frank 2000, Heinzerling and Ackerman 2002, Hwang 2016).

Discounting of future events is the CBA issue that seems to have received most attention since the 90’s, presumably as a consequence of higher academic and policy attention to the long term challenge of climate change (Azar and Sterner 1996, IPCC 1996, Nordhaus and Yang 1996,

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[10] “Alternatives are incommensurable when they cannot be precisely measured along some common cardinal scale of units of value, and incomparable when they cannot even be ranked on an ordinal scale.” Aldred, J. (2006). "Incommensurability and Monetary Valuation." Land Economics 82(2): 141-161 10.3368/le.82.2.141.

Nordhaus 2007), with a sort of peak following the publication of the Stern report on climate change (Stern 2006, Dasgupta 2007, Nordhaus 2007, Weitzman 2007, Sterner and Persson 2008). The standard techniques used to derive discount rates[11] have been shown to give discount rates so high that CBA in effect risk ignoring long term effects of decisions, which was the case with the early work by for example Nordhaus (Azar and Sterner 1996). Alternatives are discussed and one of the most prominent is hyperbolic discounting (discount rates that decline over time) (Laibson 1997, Hansen 2006, Winkler 2006, Walther 2010, Grijalva et al. 2013).

Although there are intergenerational ethical concerns with discounting, other dimensions of ethical weight are more commonly brought forward in the literature when ethical concerns with CBA are lifted. The utilitarian norms which serves as the backbone for the rationality principles of welfare economics and CBA is questioned as guiding principle and one of the most common remarks comes from egalitarian ethics (Howarth and Monahan 1996, Frank 2000, van Wee 2012). CBA is often not considering distributional effects when comparing costs and benefits and it is therefore perfectly possible that the benefit of a policy will come to individuals who are already well off, at the expense of poorer individuals. Economists have derived distributional weights to deal with the distribution problem (Drèze and Stern 1987, Adler 2016), but these weights are seldom used in applied CBAs. Another common objection is that it is morally repugnant to put a money-value on human life (Pearce 1998, Ackerman and Heinzerling 2005).

Yet another critique with an ethical basis relate to the differences between the expression of ethical values and individual preferences (Sagoff 1994), where CBA mainly focus on the latter whilst several public projects might very well appeal to the former (Holland 1996). Some thinkers have tried to reconcile this potential dissonance by stating that economic valuations only should be considered to include “restricted, preference-based accounts of welfare”, *inter alia* by ensuring that bequest values are excluded from the valuation (Adler and Posner 2006). Others have suggested that socio-economic valuations should measure effects on “experienced utility” instead of on preferences (Kahneman and Sugden 2005). Experienced utility is stated to be less sensitive to various behavioural biases common in the market place and in conventional valuation studies. If any of these two suggestions would be picked up when valuing environmental and health improvements, the subsequent CBA would be representing a more restricted sub-set of the total welfare impact of a policy. It remains to be seen if these suggestions will be picked up by the CBA community.

Of more specific interest for this thesis are the method-, and data-specific critique applicable to the methods applied to assess monetary benefits of emission reductions in Paper IV. First, there are several known (but not monetarised) benefits of emission control that remain outside the benefit analysis in Paper IV (and ECs AP-CBA). As examples, benefits of reduced acidification, eutrophication, and biodiversity loss currently are excluded. In the EC AP-CBA, also impact on climate change is excluded. Another example of omission due to lack of data is an ongoing

[11] Basically, calculating rates based on time preferences, assumptions on economic growth, and consumption elasticity of marginal utility: Ramsey discounting. Or observing preferences for monetary saving vs consumption: “average pretax rate of return on private capital”. Arrow, K. J. et al., Should Governments Use a Declining Discount Rate in Project Analysis? Review of Environmental Economics and Policy 2014, 8, (2), 145-163.

expansion of knowledge about the negative health effects of air pollution. Recently discovered negative health effects from poor air quality yet to be included in the AP-CBA includes stroke, mental health issues, diabetes, premature birth, and low birth weight (Thurston et al. 2017).

Further, and applicable to both AP-IAM and AP-CBA, there are two risks for confounding factors which would affect the estimated effect of air quality on human health. There are indications on links between premature mortality and exposure to $NO_2$ (Heroux et al. 2015), but no scientific consensus as of yet (COMEAP 2018). Further, noise pollution effects on human health also risk confounding estimated effects of air quality on human health. Current indications on risk of confounding with noise are however mixed (Stansfeld 2015, Tonne et al. 2016).

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

# 4. Potential implications of the research and future research needs

In addition to the results presented in the papers I-IV and the results directly linked to the main aim of the research leading up to this thesis, this chapter present reflections made related to the research during the PhD-student years. These relate mainly to parameters of concern when calculating emission control costs, such as data completeness, choice of climate metrics, and choice of economic perspectives. But the reflections also present perspectives on the notions of optimal emission levels and environmental taxes.

### *It is important to include all emissions affected by measures when analysing climate impacts.*

What became clear during the work leading up to Paper I is that some SLCPs usually not discussed in a climate context, such as $NO_x$, can have substantial impact on resulting cost-effectiveness calculations. But it is also clear that the silo approach predominant in air pollution and climate policy research can lead to omission of gases outside the primary concern when assessing costs and effects of control options (like omission of potential $CO_2$ effects when studying $NO_x$ options). This can have adverse effects in subsequent studies when such assumptions are less transparent. Yet again, one of the experiences drawn from Paper I is the constant reminder to retain a systems perspective, even when involved in silo policy processes.

### *Start using $CH_4$ equivalents instead of $CO_2$ equivalents when analysing SLCP options.*

Supported by the results in Paper I it can be considered that the cost-effective order of control options is largely unaffected by the climate metric used, if the measures do not affect $NO_x$ or more long-lived greenhouse gases like $CO_2$. However, the current practice of normalising the climate impact of SLCPs into $CO_2$ equivalents can cause unnecessary uncertainty and confusion due to the large variation in metric values (in Paper I, BC has metric values ranking between 4.6 and 6200 $CO_{2eq}$). To avoid this, it would be preferable if SLCP strategies and policy documents normalise SLCP emission reductions against $CH_4$ since it is more alike the other SLCPs in perturbation time than $CO_2$, as has been previously suggested by Cherubini and Tanaka (2016). In other words: compare $CH_{4eq}$ emission reduction instead of $CO_{2eq}$ emission reduction when analysing and designing cost-effective SLCP strategies. Nothing significant would happen with the relative cost effectiveness of SLCP options, but the value range of the climate metric values would become smaller and communication clearer (Table 1).

### *The choice of climate metric will affect emission levels considered cost-efficient in an AP-CBA.*

The NEC Directive policy process discussed in this thesis only included $CH_4$ at the early stage and neither costs nor benefits of $CH_4$ control was considered in the optimal model solution. However, in future policy analysis, it is probable that the climate impact of SLCP emissions will often be described by using climate metrics, as was the case in Paper IV. If so, a policy makers' perspective on climate change – expressed through choice of climate metric – will have an impact on the socio-economic optimal level of emissions.

**Table 1: Global warming potential over 100 years for the SLCPs considered in Paper I when expressed as $CH_4$ equivalents and $CO^2$ equivalents (mid values)**

| Pollutant | $GWP_{100}$ ($CH_{4\text{-eq}}$) * | $GWP_{100\text{-G}}$ ($CO_{2\text{-eq}}$) ** |
|---|---|---|
| BC | 28 | 846 |
| OC | -1.4 | -43.2 |
| NMVOC | 0.2 | 5.5 |
| $CH_4$ | 1 | 28 |
| $NO_x$ | -1 | -10.9 |

* Cherubini and Tanaka (2016)
** Paper I

This will be the consequence since most of the published guidelines on benefit assessment of avoided climate change are expressed as monetary units/$CO_{2eq}$ (United States Government Interagency working group on social cost of carbon 2013, Korzhenevych et al. 2014, Swedish Road Administration 2018). More tangible, using a recently estimated socio-economic benefit of reducing one tonne of $CO_2$ ($31/tonne (Nordhaus 2017)), the estimated monetized climate benefit of reducing BC can be anywhere between 2 024 and 99 200 $/tonne BC if policy analysis were to use easily accessible values from IPCC (147-192 200 $/tonne BC if using value ranges from Paper I). Such a range will hinder identification of one unique set of socio-economically optimal emission levels of air pollutants.

## *Where to cap a retrospective policy analysis?*

Paper II showed a partly successful proof-of-concept through the combination of decomposition analysis with qualitative analysis of implemented policies, which permitted quantification of $SO_2$ emission effects from some specific policy instruments. This concept is a step forward for policy evaluation. However, such a concept is dependent on trusting when the analysis has been meticulous enough, when the researcher can stop searching for confounding factors that would help explain emission reductions (in our case emission factor reductions). To borrow a phrase used by philosophers of science, searching for proof of an environmental policy is at risk of being subject to “experimenters’ regress” (Collins 1985), or rather in this case evaluators’ regress. An ever ongoing slightly amusing, but mostly painful, peeling of the onion layers without knowing when one has reached the core.

One example of when this risk was actualised comes from the evaluation of the eastern U.S. $SO_2$ allowance trading system that was implemented in 1995. Originally dubbed a success, later evaluations of the $SO_2$ trading showed that much of the success was due to unrelated de-regulation of the railroad rates. This de-regulation happened to enable low sulphur coal to be cost-effectively transported by train from low sulphur coal areas in the central U.S. to high sulphur coal areas in the eastern U.S. (Schmalensee and Stavins 2013). However, also this later interpretation of what it was that made the U.S. $SO_2$ emissions drop within the trading bubble can be questioned. An experienced Swedish environmental economist has as late as 2016 related that at least one U.S. individual with financial stakes in the coal power utilities lobbied for the deregulation of the railroad rates to reduce the costs of $SO_2$ emission reductions. When have one peeled enough layers?

This type of evaluation challenge with such a straightforward environmental problem as $SO_2$ emissions is an argument for caution when drawing conclusions from environmental policy instrument evaluations. Further, it can serve as a cautionary tale to the ones who advocate evidence-based environmental policies.

### *Which investment perspective to use in analysis?*

In Paper III the sometimes large effect on modelled technology choice of investment perspectives are made clearly visible. The effects could for some policy ambition levels include: which technologies considered cost-effective, which pollutants to reduce, and in which country emission reductions should take place. But what is also interesting is the implication of these perspectives when applied outside the AP-IAM model. The social planner perspective, i.e. the long-term perspective of the benevolent dictator who cares for all people and economic agents in society, is commonly used in AP policy analysis. By having this perspective in cost effectiveness analysis one can inter alia assure comparability between emitting sectors and that pure financial transactions are not part of any cost estimates (Moore et al. 2004). However, there might be a limit to the applicability of results rendered with a social planner perspective.

The corporate perspective has more similarities to actual decision making by firms than the social planner perspective, and with the results from Paper III as support it can be argued that as soon as modelled social planner cost-optimal emission levels have transformed into EU law (if not before), investment decisions starts deviating from the optimal model solution. There is a discussion in the literature on the suitability of having different discount rates between private and public projects (Grout 2003), and here it is assumed that public projects are corresponding to social planner perspectives. But accepting this notion will render a problematic situation for the analysis since the corporate perspective control cost curve is likely to look different from the social planners' control cost curve. In other words, if the optimal emission target (where MC=MB) is set with a social planner perspective, the implementation of the target might be made through non-optimal technology choices. But if the target is set with a corporate perspective, the ambition level can be suboptimal from a social perspective.

### *Which emission tax level is optimal?*

The interest rate discussion is also of concern in the discussion on policy instruments. In the literature it is argued that in the ideal market, interest rates should be equal to marginal productivity of investment and equal to marginal social time preferences (discount rate) (Marglin 1963, Azar and Sterner 1996), and this rate is identifiable with the Ramsey equation (Ramsey 1928). In fact, most often in the literature the only term used is discount rate. In extension this reasoning should imply that: if a social planner perspective is applied when evaluating the monetary value of protected ecosystems, so should it be when calculating costs of emission reductions.

When evaluating monetary values of future human health and ecosystem protection, it makes sense to have a social planner perspective given the implied long-term horizon of such a perspective. It is also conceptually straightforward to use these values as basis when setting the level of an emissions tax (since an optimal tax level should be equal to the externality). But if the social cost of emissions is calculated with a social planner interest rate and time horizon, and real-life control costs are calculated with a corporate perspective interest rate and time horizon, the emission reduction following the implementation of a tax is at risk of becoming suboptimal, at least as costs are modelled in AP-IAM. Reiterated results from Paper III can serve as a clarifying example. Assume that the social planner marginal cost of control equals the marginal benefit of control at 35 per cent of maximum policy ambition, i.e. the socio-economic optimal emission level corresponds to a 35 per cent ambition. At this ambition level, the MC of control equals around 1.3 million € for each per cent ambition for the social planner. However, for the exact same ambition level, the corporate perspective MC of control equals 4.6 million €, and the ambition level where corporate perspective MC is around 1.3 million € is at a 16 per cent ambition level. In other words, a socio-economic optimal tax rate calculated with a social planner perspective would if implemented in this example underperform with some 19 percentages of the optimal ambition level.

One can also question the theorem that in the ideal market, the discount rate should equal the interest rate, as has been done before (Persson 2008, Baumgärtner et al. 2014). One reason for separating interest rates on financial capital from discount rates on ecosystem services is because acquiring money comes at a cost. When expressed as such it is difficult to defend the notion that the discount rate of environmental benefits should equal the interest rate of investment in control technology. However, although these arguments help rationalise a differentiation in discount rates between costs and benefits in an AP-CBA, they do not solve the problems associated with the choice of economic perspective used in the analysis, as presented in Paper III.

### *Which are the key parameters for AP-CBA uncertainty analysis?*

The results in Paper IV are derived by accounting for the – given the method setup – quantifiable uncertainty in the costs and benefits of a NECA. However, the range of uncertainties that are potentially quantifiable is prohibitively large. Using Paper IV as a case for discussion, the uncertainties identified as having highest effect are the climate impact of emissions together with the economic value of climate change, as well as the economic value of avoided fatalities. However, the uncertainty analysis could also have included different trajectories of transport and fuel demand in international shipping, or uncertainty in health effects of air pollution, or uncertainty in relative fuel prices, or uncertainty in learning, or even uncertainty in precipitation patterns. And as Paper III show, also the sensitivity of interest rates on control costs could be included given the uncertainty of future market interest rates. The list can become very long, and at some point, one as a researcher must make a demarcation between uncertainties included in the analysis and uncertainties left outside. One can consider, given the importance of uncertainty in scenario studies such as in Paper III and IV, that it is time to change the terminology used when estimating uncertainty in AP-IAM and AP-CBA results. Maybe it is better to present 'model parameter variability' rather than 'uncertainty'? Better still would be to try to gather and document consensus opinions from scientists, experts, policy makers on key parameters of concern when deciding about future environmental integrity and use these for uncertainty analysis.

# 5. Final thoughts on air pollution policy support modelling

So far in this thesis the discussion has focused on the methods used in AP-IAMs and AP-CBAs and their limitations. Some proposals for improvements have been made, but also these adhere to the methods currently used. To get a more complete grasp of what type of lessons that can be learned and what long-term effect these papers can have it is important to take a wider perspective. The wider perspective can be characterised as containing facets of intra-disciplinary importance for air pollution policy support modelling, and as containing facets of science-for-policy importance. For a proper understanding of the research from these, consideration of the scientific viewpoint and general research strategy (methodology) that lies underneath these methods is needed. One must also consider the policy context and recognise that these methods are used as direct input to an active air pollution policy process and must thus adapt to the reality of policy makers. To do so the perspective of the naïve scientist operating within a well-defined environment needs to be complemented with strategic thinking and a description of the policy environment in which the methods are applied. To ensure scientific honesty it is also important to find a balance between the "freedom to doubt" necessary for scientific development (Feynman 1999, Harari 2016) and the merchandising of doubt practiced by some special interest groups (Oreskes and Conway 2011).

As has been presented, papers I-IV shows that the current methods appear robust over some model aspects but not over others. Paper V though, builds upon the already existing methodological critique of CBA presented in Åström (2017) and shows that the existing critique has gotten more support from system sciences as well as from experiment-based academic disciplines such as behavioural economics and economic psychology. Examples of these critiques are that the current AP-CBA omits technology innovation and learning, assumes perfect foresight in decision making, and disregards psychological effects of money during valuation of environmental and health effects of a policy proposal.

From this relatively large body of critique it might appear tempting to consider conventional AP-CBA as unsuitable for AP policy decision support. However, there are more aspects that needs to be considered prior to any such judgement. Given that CBA of public projects and policies is required according to legislation it is important to consider the severity and potential implications of any critique. Is the critique a call for adaptation of EU laws or adaptation of best-practice CBA within existing laws or both? Whilst not exhaustive in any way, to guide such a discussion the section below presents fundamental aspects of policy support modelling and describes the policy process within which the modelling is done.

### *Policy support models are not meant to give crystal ball predictions.*

Models, including computer models, are in principal always wrong to some extent. This should come as no surprise since every model is nothing but a nice and tidy representation of some minor part of reality. But the models, including the AP-CBA discussed in this thesis can nevertheless be improved upon. The academically top-of-mind alternative to the current AP-CBA should be to use state-of-the-art economics and modelling of the entire system driving emissions. However, the current research frontier might not be ready for adaptation into AP policy support models or to give clear policy support on integrated environmental issues. As an illustration one can make a qualitative guess on how the results from such modelling might have looked in 2013. First, consider emissions in a baseline scenario. It is likely that emissions in 2030 (17 years later than when the modelling was done) will be no higher than in 2010 due to the original NEC Directive setting emission targets to 2010 and onwards (with risk of financial penalty if transgressed). Under normal circumstances, some industrial utilities already standing in 2013 are likely to remain utilised for the remainder of their technical lifetime, which implies that a fair share remains operational by 2030. This would set a minimum level of emissions since scrapping before end of technical lifetime is often perceived as prohibitively expensive, and therefore assumed as an unavailable option. But the socio-economic system is complex. It is full of structural changes, other policy developments, ongoing revolutions in the fields if information technologies and artificial intelligence, positive technology feedbacks, innovation processes of unknown maturity, and economic decisions made under uncertainty or with conceptual biases and strategic interests. Some even refer to it as a "wicked system" (Andersson and Törnberg 2016, Andersson and Törnberg 2018). One example is the development of climate change and climate policy which makes the 'no-scrapping' assumption uncertain. Correspondingly, a state-of-the-art economic model of future air pollution emissions that accounts for all these phenomena driving emissions should have difficulty identifying an emission level between some minimum level and the NEC Directive 2010 level with higher probability than any other emission level within that range. If so, it should be difficult to sort out any difference between one future with a proposed air pollution policy and one without. But a model answer in line with: 'One cannot say with any certainty how high emissions without new policy will be in 2030, neither can one identify any changes caused by new policy' would not give support to policy makers. So, some sort of work-around is needed. One such work-around is to limit the number of phenomena included in the analysis and use rational decision rules (such as minimize costs) to constrain the state-of-the-art knowledge when applied in policy support models. Such a route can reduce the variability of outcomes so that a clear model distinction between policy and no-policy can be visualised, at the risk of losing some degree of scientific legitimacy. The current approach to AP policy support modelling can be viewed as a specific type of constrained and limited version of state-of-the-art, a type that is based on a text-book version of economic rationality.

Given the discussion above it should be self-evident that results from an AP policy support model of a potential situation in 2030 will not be actualized when the time comes, but that is not the purpose of such a model. What the modelling supporting European AP policy does, is to provide an analysis of how a multi-dimensional and logically consistent chain of events and decisions, in a well-defined environment, can imply a certain future emission level characterised by economic rationality. In this way, the AP-CBA done for the EC was like other economic models used as policy support. Rather than providing predictions (which is feasible with models of simpler physical phenomena), earlier thinkers have described economic models as providing a

> *"… level epistemological basis for debating social, political, and moral theories that can be used to frame economic policy"* (Evans (1999) on macro-economic modelling),

and present a

> *"credible counterfactual world"* (Sugden (2007) on theoretical economic models).

Instead of being evaluated against observations (of which there are none), the usefulness of this type of models have been shown to depend much on their ability to adhere to issues of direct relevance to policy makers, to remain credible through scientific rigour, and to be considered legitimate by all parties involved in the policy process (Cash et al. 2003). In brief, the most important policy requirement on this methodology and the models built upon it is logical consistency, rationality, and acceptability, not crystal ball predictions. In context of the air pollution policy support modelling discussed in this thesis these criteria are at least partly fulfilled by using knowledge from scientific working bodies in the Air Convention, by the openly available data in the models, by continuous review of the models used, and by the independency of national interests ensured when using an international institute to host the development work of the models.

### *Policy support models interacts with the policy process.*

The policy reality to which the modelling give input should also be highlighted. It is worth mentioning that the policy process partly constrains air pollution policy impact assessments such as AP-IAMs and AP-CBAs. One way in which air pollution policy impact assessments are adapted 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 measures (that also can reduce air pollution) from the analysis. This choice of system boundary can from a policy perspective be explained by the fact that responsibility for climate policy and air pollution policy in the EC is split between the Directorate-General for Climate (DG-CLIMA) and the Directorate-General for Environment (DG-ENV). DG-ENV should not propose further $CO_2$ control to the EU member states in a process outside the EU climate policy process, therefore $CO_2$ measures are not appreciated as a part of an AP-CBA for the EU. 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 or if there is a lack of consensus or lack of data, might have problems to be implemented. It is therefore unfortunate that the activities within the Network of Experts on Benefits and Economic Issues (NEBEI) of the Air Convention has been discontinued, which presumably hamper the application of state-of-the-art economics in AP policy support modelling.

Attention should be paid also to the policy experience prior to the CAPP proposal, the policy process that took place between the final AP-CBA model output, and the EU decision in June 2016. The EU policy process on updating the NEC Directive was initiated around the end period of the Air Conventions' policy process of amending the Gothenburg protocol (amended on the 4$^{th}$ of May in 2012). This Air Convention process was supported with the same tools as discussed in this thesis, but the analysis was made differently. Instead of calculating cost-efficient emission levels, the tools were used to calculate B/C ratios of possible 2020 emission reduction ambition levels. The Air Convention AP-CBA included analysis of several policy ambition levels and presented both total and incremental B/C ratios of these (Amann et al. 2011c, Holland et al. 2011). The analyses showed that benefits would exceed costs for all ambition levels by 2020, and that quasi-marginal benefits would exceed quasi-marginal costs up until a 75 per cent ambition level (50 per cent for ozone damages). Despite these results that advocated more efforts to reduce emissions, the policy makers finally opted for emission levels that corresponded to a <0 per cent modelled ambition level in the 2012 amendment of the Gothenburg protocol.

At first, also the EC process of preparing for a CAPP proposal utilised the same approach as was used in the Air Convention process, but during 2012/2013 the approach changed. By March 2013 the policy support material had changed from using predetermined policy ambition levels as basis into letting models identify cost-efficient (albeit restricted) ambition levels with adherence to economic rationality (Amann et al. 2013). The models identified that by 2025, the cost-efficient ambition level corresponded to 75 per cent of the technically maximum ambition level, but results were not presented for the costs and benefits of pathways towards, or achievement of, the already established long-term policy ambition of the 6th and 7th EU environment action programme for air pollution:

> *"levels of air quality that do not give rise to significant negative impacts on, and risks to human health and environment."* (European Union 2013).

Prior to proposing CAPP and an updated NEC Directive on the 18th December 2013, internal discussions at the EC as well as external pressure (Harmsen and van Vilsteren 2016), had lowered the ambition level (especially for $NH_3$) and the target year had been shifted to 2030, so the proposed 2030 target corresponded to a 67 per cent ambition level. During the EU negotiations that followed at the EU council and parliament the policy process continued to reduce the ambition level. And as has been presented earlier, the EC even considered to scrap the revision completely. But despite lowered ambition and a push from the mayors of London and Paris (DW 2016), no agreement was reached on the last Environmental council on the 20th of June (European Council 2016b). However, despite the fact that no more meetings on the issue could be held, and much thanks to active footwork by the Dutch, on the 30th of June 2016 (the very last day of the Dutch EU presidency) the council and parliament reached a provisional agreement on an updated NEC Directive (European Council 2016a). The objective of the updated NEC Directive is that human health impacts in 2030 from air pollution should be 50 per cent of those in 2005. This corresponds to a 40 per cent model policy ambition level in 2030 given that the objectives of the recently adopted EU climate and energy policy is achieved (54 per cent ambition level if not considering the EU climate and energy policy). This can then be compared to the <0 per cent ambition level in 2020 reached in the Air Convention four years earlier.

All in all, it can be settled (presumably yet again) that it is an impossibility to require truthful predictions from models used for future studies of unmeasurable parameters, and democratic policy processes are likely to result in political ambition levels different from modelled ambition levels. Further, it can be argued that the new approach encouraged higher policy ambitions. But it can also be argued that the approach implied utilization of CBA beyond its scope of application. One example is that the AP-CBA presented did not compare alternative policy ambitions even though policy ambitions were available (as well as alternative scenarios) whilst the main purpose of CBA is to compare options. Also, much economic knowledge is yet to be incorporated into the models, and only one type of economic rationality was considered. The approach might have been the best available, but improvements are still needed.

### *So, what are the alternatives?*

There are of course alternatives to the current policy support modelling. There is always the alternative to increase current efforts to improve data and trim the existing methods, as is already done in the atmospheric dispersion parts of the AP-IAM modelling (Kiesewetter et al. 2013, 2014, 2015a, 2015b, Wagner et al. 2018). As is presented in Paper V, there are data gaps of economic relevance in the existing methods and improving the data would give better support. Further proposals for method improvements are already available in the literature, and these should be evaluated with respect to logical consistency, rationality, and acceptability. Examples of proposed method developments that can be the starting point for evaluation are Extended CBA (Holland et al. 2005) which allows for inclusion of non-quantified data, and the similar Qualitative CBA (van den Bergh 2004) which could enable consideration of the precautionary principle. Other examples are the use of Dynamic CBA´s which links economic equilibrium models to CBA (Kriström and Bonta Bergman 2014), or CBA based on experimental and behavioural economics (Gowdy 2004, Gowdy 2007, Brennan 2014). A minimum ambition of data and method development should be to have CBA models that can model fulfilment of already existing long-term policy ambitions, such as the EUs environmental action programme for air pollution, despite differences in target years. Continuous improvements of data and methods can imply that current legal obligations on socio-economic impact assessments of policy proposals needs to encourage continuous data and method adjustments in accordance with state-of-art CBA research. Further, the proposed minimum ambition level implies that policy makers and scientists, if they have not done so already, will have to update which air quality levels that corresponds to the long-term ambition. Given that the highest current activity in CBA method development seems to be found in climate economics and health economics, AP-CBA practitioners could increase outreach efforts to these communities to share knowledge and expertise and increase the amount of data available.

Another important improvement would be to use AP-CBA to compare more alternatives than just a baseline scenario and a cost-efficient solution. This type of use is after all the intended use of CBA advocated by environmental economists. As a reminder, ‘Specification of the alternatives to be evaluated’ is the first item when doing a text-book CBA (Boardman et al. 2001). More policy alternatives will however require more effort from the EC, an increase in effort that can be recommended.

Yet another variant of improving on, or rather increasing modesty of, current methods is to utilise the uncertainty analysis already made by modellers when producing impact assessments. Both the GAINS model and the ARP model used to give AP-CBA policy support have over the years shown interim results from uncertainty analysis over key parameters. These sensitivity results could be combined, reiterated, weighted and included into the final impact assessment delivered to policy makers. One can also alter the communication of AP-CBA results when including the results from an uncertainty analysis in the main results to better account for the limitations and uncertainties. Fortunately, CBA results can be represented in a simplifying way by putting it into answering a question with ‘yes’ or ‘no’ since the questions asked in a CBA can be interpreted in form of: “Is policy A preferable to policy B?”, or “Is X amount of emissions socio-economic optimal?”. Further, the IPCC has to some extent established a terminology for communication of uncertainty (IPCC 2005). And even though finetuning might still be necessary

(Budescu et al. 2014), such established terminology can be used also in AP-CBA. If so, the answer from a CBA on two alternative air pollution policies could be communicated as:
'Yes, it is very likely that option A is preferable to option B.', or
'Yes, an emission reduction of at least X per cent is likely to be socio-economic optimal."
Based on the review in Paper V it can be argued that such communication of CBA results is better aligned with current understanding of the methodological challenges of the CBA concept and the inherent uncertainties of scenario studies.

Large research efforts are needed to include more aspects of the complex socio-technic-economic-environmental system into air pollution policy support models, efforts that would make our understanding and policy support models clearer, and existing state-of-the-art science converted to formats more useful to policy makers. Such efforts require corresponding research funds made available. But money for fundamental economic research is in short supply for scientists studying European air pollution policies. Given lack of enough research funds, the AP policy research community would probably have to rely on the success of other research groups that tries to push the scenario modelling research forward, and the recommendations to create a platform of scenario models, made by the EC expert group on foresight modelling (Köhler et al. 2015), can be supported. As a bare minimum, given the growing body of indications on methodological shortcomings of AP-CBA summarised in for example Åström (2017) and Paper V, research efforts must be made to clarify which (if any) of the known economic phenomena currently omitted in the policy support modelling that would have substantial effect on the model results if included. Examples of phenomena that can be evaluated are technology learning (Höglund-Isaksson et al. 2016) and perfect foresight. Clarifying potential impacts of these phenomena is needed to increase scientific legitimacy of the policy support models.

As has been presented, the AP-CBA used by the EC is constrained and assumes a text-book version of economic behaviour, a behaviour described by Amartya K. Sen (1977) as belonging to "rational fools". Furthermore, economic analysis is not neutral with respect to moral and ethical values (Evans 1999). Correspondingly there are other equally realistic assumptions and constraints to be considered, and there are other types of rational reasoning that can be utilized. As a reminder from the discipline of cultural theory, different classes of co-existing rationalities in society, such as 'egalitarian', 'hierarchical', and 'individual', has been suggested for integrated assessments (Thompson 1997). Studies suggest that these help explain how environmental issues can be perceived different among different groups of society (Steg and Sievers 2000), and these different rationales are since long represented in another environmental policy support discipline: life cycle analysis (Hofstetter et al. 2000, Huijbregts et al. 2017). In other words, a fifth alternative to the current approach is to also represent other forms of rationality and corresponding models and deliver results from these to the AP policy makers in addition to the AP-CBA results. The market place for clean air is not the only rational perspective available.

As a simple example, emission-only models can be used to find emission levels in 2030 that meets the objectives of the EUs environmental action plan. Such analysis would provide a complementary rationale to the cost-effectiveness rational of AP-IAM models and the cost-efficiency rationale of AP-CBA models, and not be constrained by the data availability and methods of current AP-IAM and AP-CBA models. An additional example is that the cost-effectiveness rationale of AP-IAM could be complemented with equity-based rationales in analysis of emission control costs. One could for example adjust national costs of emission control technologies with national estimates of marginal utility of consumption (Evans 2005). Such analysis would render a cost-effective solution adjusted for cost-equity within the European region. To follow the equity analysis path, the AP-CBA could weight health and environmental improvements higher in countries with low baseline life expectancy. This could be done by adjusting the constraints on years of life lost in the cost optimization with a factor corresponding to each countries difference from the European mean life expectancy.

The next European air pollution policy initiative is likely the 3rd revision of the Air Convention Gothenburg protocol, potentially also a revision of the EU Air Quality Directive. If the policy support used for this revision adds any of the suggestions listed above, it would be an improvement compared to the policy support produced for the 2013 proposal of CAPP and revision of the NEC Directive. If implemented, these suggestions can provide policy makers with a larger set of policy support material, and the support will be more responsive to existing political priorities whilst ensuring theoretical consistency and robustness of models and methodology. As a minimum, the policy support should add at least an AP-CBA model that can solve the emission levels necessary to reach the EU action plan targets and add comparison of more alternative policy solutions. Even if this implies extension of the requirements on EU impact assessments, as well as an extended knowledge-based co-ordination of air and climate policies within EU and other international bodies.

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