JONATHAN VAN DER KAMP

SOCIAL COST-BENEFIT ANALYSIS OF AIR POLLUTION CONTROL MEASURES

Advancing environmental-economic assessment methods to evaluate industrial point emission sources

KIT Scientific Publishing

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by

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Dissertation, Karlsruher Institut für Technologie
KIT-Fakultät für Wirtschaftswissenschaften

Tag der mündlichen Prüfung: 03. Mai 2017
Gutachter: Prof. Dr. rer. nat. Ute Karl
Prof. Dr. rer. pol. Frank Schultmann

Impressum

KIT Scientific Publishing

Karlsruher Institut für Technologie (KIT)
KIT Scientific Publishing
Straße am Forum 2
D-76131 Karlsruhe

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www.ksp.kit.edu

Print on Demand 2017 – Gedruckt auf FSC-zertifiziertem Papier

ISBN 978-3-7315-0697-3
DOI 10.5445/KSP/1000072046

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zur Erlangung des akademischen Grades eines

Doktors der Wirtschaftswissenschaften (Dr. rer. pol.)

von der Fakultät für Wirtschaftswissenschaften
des Karlsruher Instituts für Technologie (KIT)

genehmigte

Dissertation

von

Dipl.-Wi.-Ing. Jonathan van der Kamp

aus Karlsruhe

Tag der mündlichen Prüfung: 03. Mai 2017
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Kurzfassung

Luftschadstoffe erhöhen das Risiko für Atemwegs- und Herz-Kreislauf-Erkrankungen sowie das Sterblichkeitsrisiko der Bevölkerung und führen somit zu gesellschaftlichen Kosten. Wohlfahrtsorientierte Politik zielt darauf ab, diese Kosten in Marktpreise zu integrieren, z.B. durch Grenzwerte für industrielle Luftschadstoffemissionen. Gemäß der neoklassischen ökonomischen Theorie sollten Emissionen verringert werden, bis die damit verbundenen Grenzkosten und -nutzen gleich groß sind. Dieses Gleichgewicht kann durch die gesellschaftliche Kosten-Nutzen-Analyse (KNA) bestimmt werden, welche auf eine effiziente Verteilung knapper Ressourcen und die Vermeidung unverhältnismäßiger Kosten abzielt. Hierdurch wird die KNA auch für betroffene Industriesektoren interessant.

Ansätze für die gesellschaftliche KNA von Emissionsminderungsmaßnahmen sind im politischen Entscheidungskontext weit verbreitet, da entsprechende Analysen verpflichtend zur Vorbereitung neuer Gesetze sind. Ansätze zur Entscheidungsunterstützung auf der betrieblichen Standortebene sind jedoch rar. Existierende Bewertungsmodelle erfassen die zeitliche und räumliche Variabilität von Eingangsdaten nur unzureichend.

Die vorliegende Arbeit beabsichtigt diese Forschungslücke durch methodische Weiterentwicklungen für die gesellschaftliche KNA von Emissionsminderungsmaßnahmen zu schließen. Der Fokus liegt auf der Bewertung von Gesundheitsschäden. Zentrale Zielsetzungen sind der Transfer der gesellschaftlichen KNA auf die Standortebene, die Analyse des Einflusses von Modell- und Methodenparametern auf die Ergebnisse und die Ableitung methodischer Empfehlungen für private Entscheidungsträger unter Berücksichtigung von Unsicherheiten.

Hierzu wird der sogenannte Wirkungspfadansatz detailliert untersucht und mittels eines neuen methodischen Ansatzes zur Gesundheitsschadensbewertung implementiert. Dieser beruht auf der Einbindung von zeitlich und räumlich hoch aufgelösten Schadstoffausbreitungsrechnungen. Für die gesellschaftliche KNA wird die Gesundheitsschadensbewertung mit einem betriebswirtschaftlichen Ansatz zur Abschätzung der privaten Emissionsminderungskosten kombiniert.

Die zeitliche und räumliche Modellauflösung, in Verbindung mit Emissionsquellcharakteristiken und der Bevölkerungsdichte, werden durch Sensitivitätsanalysen als zentrale, ortsspezifische Einflussfaktoren auf die Gesundheitsschäden identifiziert. Die Berücksichtigung NO_2-expositionsbezogener Sterblichkeit führt zu einer deutlichen

Erhöhung der Schäden in Metropolregionen, hängt jedoch vom Konzentrationsschwellwert, der Emissionsquelle und der Modellauflösung ab. Der größte Teil der quantifizierten Schäden betrifft Sterblichkeitsrisiken durch eine langfristige Feinstaubexposition, weshalb damit verbundene methodische Aspekte besonders einflussreich sind.

Die neu entwickelte Methodik verringert Unsicherheiten, stellt jedoch hohe Ressourcenanforderungen. Das vertretbare Unsicherheitsniveau und der zu bevorzugende methodische Ansatz richten sich somit nach dem Entscheidungskontext. Die vorliegende Arbeit zeigt, dass Sensitivitätsanalysen die Belastbarkeit der Ergebnisse erhöhen. Die Ergebnisse unterstreichen weiterhin die Bedeutung methodischer Leitlinien, um Willkür bei der gesellschaftlichen KNA zu vermeiden und somit die Berücksichtigung von Gesundheitsschadenskosten in Entscheidungssituationen zu erleichtern.

Abstract

Exposure to ambient air pollution increases the risk for humans of developing respiratory or cardiovascular diseases as well as of dying prematurely, thus imposing costs on society. Welfare-oriented policy-making aims at integrating these costs into market prices, e.g. by reducing harmful atmospheric emissions from industrial sources. Following neoclassical economic theory, emissions should be reduced up to the point where the marginal costs equal the marginal benefits of emission control. Social cost-benefit analysis (CBA) is a method that serves to approximate this equilibrium, aiming at spending scarce resources efficiently and avoiding disproportionate costs. This principle also bears an interest for concerned industry sectors, as it can under certain conditions be used to apply for exemptions from too stringent regulations.

Scientific methods for social CBA related to air pollution at an aggregate level are well developed in the public policy context, as they are a mandatory support for the preparation of new regulations. Yet, approaches for decision-support at the level of industrial point emission sources are rare. Existing models do not properly capture temporal variations in emission patterns and spatial variations in population density.

This thesis aims at filling this gap by enhancing economic methods for social CBA of emission control measures, notably regarding health damage cost assessment. The main objectives are to transfer the social CBA method to the site level, test the influence of key modelling and methodological choices on health damage costs, and derive recommendations with regard to uncertainty and private sector decision-making.

To achieve these objectives, the so-called impact pathway approach is comprehensively reviewed and implemented into a new modelling framework for health damage cost assessment of classical air pollutants. It makes use of air pollutant dispersion modelling of high temporal and spatial resolution. For the purpose of social CBA, health damage cost assessment is combined with a private cost characterisation method.

Case study results confirm that temporal and spatial atmospheric modelling resolution, in conjunction with emission source characteristics and population distribution, are key site-dependent influencing factors on health damage costs. Including NO_2 exposure-related mortality considerably increases results in metropolitan areas, however depending on further factors such as concentration threshold, emission source characteristics, and modelling resolution. The largest share of health damage costs is linked to premature mortality from long-term $PM_{2.5}$ exposure, making related methodological choices particularly influential.

The newly developed approach decreases uncertainty, however at the expense of considerable resource requirements. The tolerable level of uncertainty and the preferred assessment approach thus depend on the decision context. This thesis shows that sensitivity analyses of key influencing factors increase the robustness of results. The results underline the importance of methodological guidance from official bodies to avoid arbitrary choices in social CBA, thus facilitating the integration of health damage costs into decision-making.

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Abbreviations

AAQD	Ambient Air Quality Directive (Directive 2008/50/EC)
BAT	Best Available Techniques
BREF	BAT REFerence (document)
CAFE	Clean Air For Europe
CAPEX	Capital Expenditures
CBA	Cost-Benefit Analysis
CCS	Carbon Capture and Storage
CH4	Methane
CLRTAP	Convention on Long-Range Transboundary Air Pollution
CO2	Carbon dioxide
CRF	Concentration-Response Function
CVM	Contingent Valuation Method
DeNOx	De-Nitrification (system for reducing NOx emissions)
DPSIR	Driver-Pressure-State-Impact-Response
EAP	Environmental Action Programme
EEA	European Environment Agency
EGTEI	Expert Group on Techno-Economic Issues
EIAD	Environmental Impact Assessment Directive (Directive 2014/52/EU)
ELV	Emission Limit Value
EMEP	European Monitoring and Evaluation Programme

ESP	Electrostatic Precipitator
ETS	Emission Trading Scheme
EU	European Union
ExternE	Externalities of Energy (EU research project series)
FR	France
GAINS	Greenhouse Gas and Air Pollution Interactions and Synergies
GDP	Gross Domestic Product
GHG	Greenhouse Gas(es)
GIS	Geographic Information System
HEIMTSA	Health and Environment Integrated Methodology and Toolbox for Scenario Assessment (EU research project)
HRAPIE	Health Risks of Air Pollution In Europe
IDF	Île-de-France
IED	Industrial Emissions Directive (Directive 2010/75/EU)
IPA	Impact Pathway Approach
IPPC	Integrated Pollution Prevention and Control (Directive 2001/80/EC)
LAU	Local Administrative Units
LCOE	Levelized Cost Of Energy
LNB	Low NOx Burner
MAC	Marginal Abatement Cost (approach)
MCPD	Medium Combustion Plants Directive (Directive EU/2015/2193)
MDC	Marginal Damage Cost (approach)
MRAD	Minor Restricted Activity Days

MW(h)	Megawatt (hour)
NECD	National Emission Ceilings Directive (Directive 2016/2284/EU)
NEEDS	New Energy Externalities Developments for Sustainability (EU research project)
NewExt	New Elements for the Assessment of External Costs from Energy Technologies (EU research project)
NFR	Nomenclature For Reporting
NH3	Ammonia
NMVOC	Non-Methane Volatile Organic Compounds
NO_x/NO_2	Nitrogen oxide/Nitrogen dioxide
NUTS	Nomenclature of Territorial Units for Statistics
OPEX	Operational Expenditures
PIG	Plume-in-Grid
PM	Particulate Matter
$PM_{2.5}/_{10}$	Particulate Matter with an aerodynamic diameter below 2.5/10 µm
RAD	Restricted Activity Days
SCR	Selective Catalytic Reduction
SIA	Secondary Inorganic Aerosols
SIREAM	SIze REsolved Aerosol Mode
SO2	Sulphur dioxide
SOMO35	Sum Of Means Over (35 parts per billion)
TFTEI	Task Force on Techno-Economic Issues
TSAP	(EU) Thematic Strategy on Air Pollution

UNECE	United Nations Economic Commission for Europe
VOLY	Value Of a Life Year
VSL	Value of a Statistical Life
WACC	Weighted Average Cost of Capital
WHO	World Health Organization
WLD	Work Loss Days
WRF	Weather Research and Forecasting (model)
YOLL	Years Of Life Lost

Preface

Publishing this thesis has taken several years and involved lessons on scientific rigour, collaborative work, intrinsic motivation etc. Having finished this process (but not the learning), I would like to thank several people and institutions for their valuable support:

My supervisor Prof. Ute Karl shaped my early professional development. Her experience helped me to distinguish the academic from the business world, which is particularly useful when transferring academic methods to a business context. Many thanks also to my co-supervisor Prof. Frank Schultmann (and the Institute for Industrial Production (IIP) at the KIT) for the assistance and for giving me the opportunity to learn from and exchange with academic peers. With Till Bachmann I share fruitful working years - through his profound scientific background he provided valuable support to this thesis and beyond. Robin Lorenz did a great job in supporting me with programming tasks. Thanks as well to my collaboration partners at CEREA for their contribution and, of course, to EIFER, including my colleagues, for the productive atmosphere that enabled me to pursue this thesis. Last but not least, I would like to mention my family: apart from gently pushing for progress from time to time, what matters most is their mere existence.

I gratefully acknowledge support from the Karlsruhe House of Young Scientists which enabled me to broaden my horizon during a research period in Singapore. The Stiftung Landesbank Baden-Württemberg generously supported the publication of this thesis.

Finally, I would like to express some motives underlying this thesis. The fact of being a business engineer partly explains the contents (i.e. a bit of everything, aka a transversal and multidisciplinary work). The topic is founded in my conviction that our environment should be preserved. Accordingly, I oriented my academic career towards energy (a life supporting good but also cause of environmental harm) and environmental economics. There are certainly more obvious and effective ways of protecting the environment than writing a thesis that aims at establishing environmental economic methods in an industrial context. Yet, industry should be part of the solution. Advancing scientific methods that aim at spending scarce resources efficiently, whilst at the same time integrating the environment into decision-making is thus my contribution. Though acknowledging limitations related to these methods, I consider that applying and discussing them in a sense of continuous improvement is globally more helpful than rejecting them.

Karlsruhe, June 2017

1 Introduction

1.1 Context and problem definition

A welfare optimizing allocation of resources in markets is often inhibited by the existence of so-called externalities or, when expressed in monetary terms, external costs. External costs comprise those positive or negative effects related to producing a good or delivering a service which are not reflected in its market price. A typical example of externalities and a key element henceforth are unpriced human health impacts caused by atmospheric emissions that originate from fossil fuel combustion, e.g. in the public power or transport sector. Another prominent example related to combustion processes, though not at the centre of this thesis, are anthropogenic greenhouse gas emissions that contribute to global warming (IPCC 2014b).

Policy-making concerned with reducing market distortions and increasing welfare aims at internalising external costs. This involves their integration into market prices through incentive-based instruments or quality standards. In doing so and with regard to environmental policy-making, the European Union (EU) applies the "polluter pays principle", requiring that polluters take financial responsibility for the damage caused by their activities (article 191 (2), consolidated version of the Treaty on the Functioning of the European Union, 2012). Moreover, the EU demands that "*available scientific and technical data*" as well as "*the potential benefits and costs of action or lack of action*" are accounted for in environmental policy-making (article 191 (3), ibid.). This points to the need to balance environmental with wider economic indicators, e.g. by simultaneously considering the private costs and societal benefits (i.e. avoided damages) related to air pollution control measures. To fulfil this task, reliable science-based methods are required, enabling policymakers to

1) assess air pollution-related external costs or benefits, and

2) put external costs into perspective with private effects.

These methods are the centrepiece of this thesis. Their respective state of scientific advancement shall be briefly summarised in order to motivate the methodological developments carried out in the following.

Despite early pioneering work by the economist Pigou (1912) and first environmental-economic approaches related to energy emerging in the 1980s (Hohmeyer 1988, Mendelsohn 1980), it was only in the 1990s that the first comprehensive methods for the monetary assessment of externalities in the energy sector were developed in a joint US-European research initiative (European Commission 1995, Oak Ridge National Laboratory and Resources for the Future 1992). A great deal of work was devoted to human health impacts caused by atmospheric emissions that proved to represent a substantial share of quantifiable damages and that are still a cause of concern. Since then, corresponding assessment methods have evolved continuously, reflecting updated scientific evidence and improved modelling capabilities (cf. chapter 4).

Yet, when aiming to assess health impacts caused by individual fossil fuel power plants, more variable operation patterns, induced by recent market developments (cf. section 2.3), require the corresponding assessment methods to be adapted. Most existing European-wide assessment models fail to properly account for input data variability, since they are based on a roughly resolved and averaged modelling in terms of space and time (cf. sections 3.5 and 5.5.2). This neither allows assessing spatial variations in population exposure properly, nor does it account for temporal variations in emissions and meteorological background conditions during the year. Using a highly-resolved space- and time-specific atmospheric modelling is thus identified as a way of improving the health impact assessment of industrial point emission sources (cf. chapter 6).

Social cost-benefit analysis (CBA) is frequently applied to compare private and external effects, particularly in the public policy context. In the energy sector and with regard to air quality, social CBA has been regularly used as a support to define appropriate levels of policy intervention at an aggregated level, e.g. the country or European level. The overall efficiency of policy scenarios is assessed, essentially by balancing private emission control costs and societal health (or other) benefits from reduced pollution levels. Whilst acknowledging some limitations of CBA (cf. section 8.4), its use in the regulatory context is nowadays well accepted. Applications of social CBA at the level of a particular point emission source are rare (cf. section 3.5). However, through recent policy changes with regard to industrial emissions (cf. section 2.4.6) and more generally as a means to foresee and cope with increasing environmental requirements, the use of CBA presents opportunities for the private sector. The social CBA methodology therefore needs to be transferred from the public policy domain to the business domain, which is another key undertaking of the current thesis.

As a first step towards this objective, an exploratory social CBA of emission control at a coal-fired power plant is conducted in chapter 5, focusing on variability in damage costs due to geographical settings. However, the employed model uses partly outdated input data and, most critically, does not account for increasingly variable operation profiles. Other existing models rely likewise on averaged damage cost factors. This lack of a social CBA methodology applicable to industrial point emission sources is tackled by the dedicated methodological advancements of this thesis (cf. chapter 6). Even though these developments focus on the energy sector, the general approach to social CBA can be easily transferred to other industrial sectors concerned with emission control measures.

Through practical case studies within this thesis, new insights are gained on the influence of modelling features and methodological choices on health damage costs. These insights feed into a thorough discussion of uncertainty underlying social CBA and health damage costs in particular. To conclude the methodological developments and based on the insights generated within this thesis, specific recommendations for the application of social CBA in the business context are derived (cf. section 8.5).

1.2 Goal definition

The overarching goals and corresponding sub-objectives of this work are to:

- Extend the existing scientific methods for the assessment of human health damage costs caused by atmospheric emissions from fossil fuel power plants in order to cope with temporal variability in operation profiles and spatial variability in population densities. This shall be achieved by developing a new assessment framework, building upon a highly resolved atmospheric modelling.

- Transfer the methodology for social cost-benefit analysis of emission control measures from the public policy to the private business context. This shall be achieved by

    - combining the health damage cost approach with a private cost characterisation approach in order to carry out social CBAs of emission control measures at site level;

    - transparently evaluating the underlying approaches, their strengths and limitations.

- Derive methodological recommendations accounting for uncertainty in health damage costs. This shall be achieved by
    - varying the most influential parameters regarding modelling features and methodological choices in order to analyse the sensitivity of results with respect to key parameters;
    - discussing uncertainty on a quantitative and qualitative basis and describing the consequences of uncertainty for public and private decision-making.

1.3 Approach and related contents

Figure 1.1 summarises the global approach followed in this thesis as well as related contents.

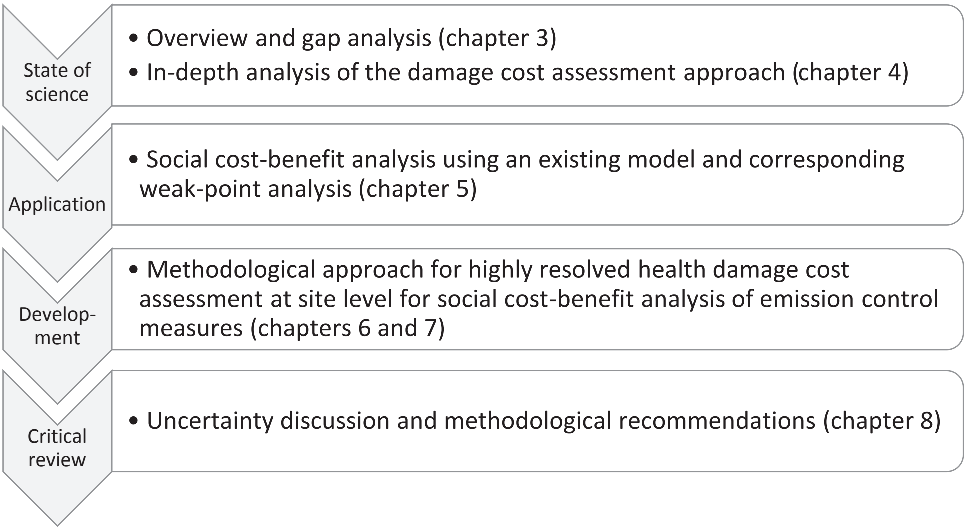

Figure 1.1: General approach and underlying methodological developments of the current thesis

Following the above depicted approach, this thesis includes the following contents, each chapter ending respectively with a short summary:

In chapter 2, the current economic and environmental challenges of fossil fuel-fired power generation in Europe are described. This notably includes environmental and health impacts caused by atmospheric emissions and the associated regulatory framework, providing the rationale for economic assessment of emission control measures.

In chapter 3 the key methods underlying social CBA are presented, split into techno-economic and environmental-economic approaches. After introducing the theory, a comprehensive literature overview serves to categorise existing scientific work dealing with the economic assessments of emission reduction measures at fossil fuel power plants. Crucially, this overview points out the limitations of currently available models with regard to variable power plant operation and therefore motivates the development of a new assessment framework within this thesis.

In chapter 4, an in-depth methodology analysis and review of implementations of the damage cost approach is carried out. The aim is twofold. On the one hand, the methodological evolution over the past 20 years is transparently described and illustrated, based on existing and newly derived damage cost estimates. On the other hand, quantitative analyses are carried out regarding the influence of key parameters and assumptions on health damage costs, providing information for sensitivity analysis.

Chapter 5 features an application of social CBA at an exemplary power plant using an existing assessment model. The analysis demonstrates the influence of the geographic emission source location on quantified damages amongst other factors. An in-depth gap analysis of the assessment model used reveals its limitations, e.g. the use of time- and space averaged atmospheric modelling results.

A new methodological framework for health damage cost assessment at site level is developed and implemented in order to tackle the key limitation of existing models, as presented throughout chapter 6. This framework is completed by developing the second key element needed for social CBA, i.e. a costing methodology for selected primary and secondary emission control measures.

In chapter 7, the newly developed damage cost assessment framework and the costing methodology are applied in several case studies that are classified into three categories, each serving to analyse specific aspects. The influence of emission patterns and atmospheric modelling features on health damage costs is analysed first, permitting a quantitative comparison of a highly resolved modelling to time- and space-averaged modelling.

This is followed by an analysis of methodological assumptions concerning health impact assessment. Finally, social CBAs of emission control measures at site level are carried out, discerning primary and secondary abatement measures. Sensitivity analyses test the influence of key parameter and methodological assumptions on the results obtained, providing new insights on uncertainty and the overall robustness of the developed framework.

Based on the results of the previous chapters, chapter 8 starts with a discussion of the case study results. Moreover, it includes a comprehensive analysis of uncertainty underlying health damage costs and a discussion of how uncertainty influences decision-making in the private and policy context. This is followed by a critical appraisal of the developed methodology, serving at the same time to identify future research opportunities. Methodological recommendations regarding the application of the social CBA method in the business context conclude the chapter.

In chapter 9 the contents of this thesis are summarised, followed by an outlook.

2 Background: fossil fuel-fired power generation under changing economic and environmental conditions

This chapter describes the larger context of fossil fuel-fired power generation in Europe, setting the scene for the scientific undertakings in this work. The overarching European climate and energy policy targets are presented, as they are important drivers for the future development of the energy system including the future of fossil fuel-based power generation (cf. section 2.1).

Traditional strengths but also weaknesses regarding fossil fuel-based power generation are discussed with special regard to the often-cited triangle of objectives: security of supply, competitiveness, and environmental soundness (cf. sections 2.2 to 2.4). Particular attention is given to the environmental impacts caused by fossil fuel power plants, as they are a key element in the methodological developments of this thesis.

Due to the transboundary nature of atmospheric substance dispersion and the predominant influence of European energy and environment policy, the current chapter takes a European perspective in the sense that most background information, statistics, and results will be presented at the aggregated European Union (EU) level. This is partly complemented by national data for France and Germany.

2.1 The European climate and energy policy framework

The short- and long-term development of energy markets is strongly defined by overarching political targets, especially with regard to greenhouse gas (GHG) emission reductions, the share of renewable energy, and energy efficiency objectives (Table 2.1). As indicated, the degree of commitment varies between these targets and between years, leading to more or less emphasis with which these targets are pursued in practice.

Table 2.1: European (EU), German (DE) and French (FR) energy policy targets in 2020, 2030 and 2050 (Assemblée Nationale 2015, BMBU 2016, BMWi 2014, Bundesregierung 2010, European Commission 2008a, 2011, 2014b, 2015b, 2016, European Parliament and Council of the European Union 2009)

		2020	2030	2050
GHG emission reduction (towards the base year 1990)		EU: 20 % [b] DE: 40 % [b] FR: 31 % [b]	EU: 40 % [b*] DE: 55 % [nt] FR: 40 % [nt]	EU: 80 - 95 % [n-b] DE: 80 - 95 % [nt] FR: 75 % [nt]
Renewable energy share in gross final energy consumption		EU: 20 % [b] DE: 18 % [b] FR: 23 % [b]	EU: 27 % [b*] DE: 30 % [nt] FR: 32 % [nt]	EU: not available DE: 60 % [nt] FR: not available
Energy efficiency	**Primary energy consumption reduction**	EU: 20 % [n-b*] (base year 1990) DE: 20 % [n-b*] (base year 2008)	EU: 30 % [b*] (base year 2007) DE: not available	EU: not available DE: 50 % [nt] (base year 2008)
	Final energy consumption reduction	FR: 17.4 % [n-b*] (towards a 2020 baseline scenario)	FR: 20 % [nt] (base year 2012)	FR: 50 % [nt] (base year 2012)

[b] = binding target; [b*] = proposed binding target; [n-b] = non-binding target; [n-b*] = non-binding overall target but with binding obligations at national level; [nt] = national target

For the year 2020, binding GHG and renewable energy targets were first set at the EU level and then broken down into binding national commitments. The European Commission can therefore start infringement procedures in the case of non-respect of these targets. For the year 2030, the European Commission has decided to set an average minimum European greenhouse gas reduction target without prescribing concrete and binding national targets. Likewise a minimum share of renewable energy consumption at European level is defined without a national breakdown (European Commission 2014b). Recently proposed legislation would also imply a binding energy efficiency target at EU level for 2030 (European Commission 2016). Although France and Germany (like other member states) have defined their own national targets, for the time being these are neither binding towards the European Commission nor binding at national level as they may be adapted by future governments. In the long run, i.e. up to 2050, certain targets are already specified, without however having a binding status so far. While the targets for greenhouse gas reduction and renewable energy shares are clearly defined, the case is more complex for energy efficiency, where targets are either expressed in absolute or relative terms and with regard to primary or final energy consumption. In addition, national targets depend on further factors, e.g. whether certain sectors such as air transport are included or whether a temperature normalisation of consumption data is carried out. This makes cross country comparisons difficult.

The influence of these policy targets (and further influencing factors) on the future perspectives of fossil fuel-fired power generation are discussed next.

2.2 Historic strengths of fossil fuel-fired power generation

Fossil fuel-fired power generation, largely based on burning coal (including lignite), oil or natural gas in large combustion plants, has played an important role in the European energy mix for many decades. In the past, its main advantages over other forms of electricity generation were availability, henceforth termed security of supply, and competitiveness.

2.2.1 Security of supply

"Security of supply", i.e. the availability of electric power in sufficient quantity and quality from traditionally large centralised power plants, is a main advantage of fossil fuel-fired power generation compared to other generation technologies. Fossil fuel-fired power plants are "dispatchable", meaning their operation profile can be adjusted to match the demand side, both in terms of electricity supply and auxiliary services, e.g. frequency regulation (Joskow 2011). Their degree of flexibility in terms of minimum load, minimum operating time or ramp up -and down rates is subject to technical constraints and differs between power plant types (Brauner et al. 2012).

Not only in the short-term but also in the long run, fossil fuel-fired power generation has the capacity to secure the supply of electricity, due to substantial worldwide fuel reserves. While emphasizing the uncertainties inherent in any kind of future projection, a comparison of global reserves (i.e. "*proven volumes of energy resources economically exploitable at today's prices and using today's technology*") and resources (i.e. *"proven amounts of energy resources which cannot currently be exploited for technical and/or economic reasons, as well as unproven but geologically possible energy resources which may be exploitable in future"*) with projected consumption data up to 2040 (Figure 2.1) indicates that fossil energy carriers remain available in abundant quantity for several decades or even centuries to come (BGR 2015, 2016).

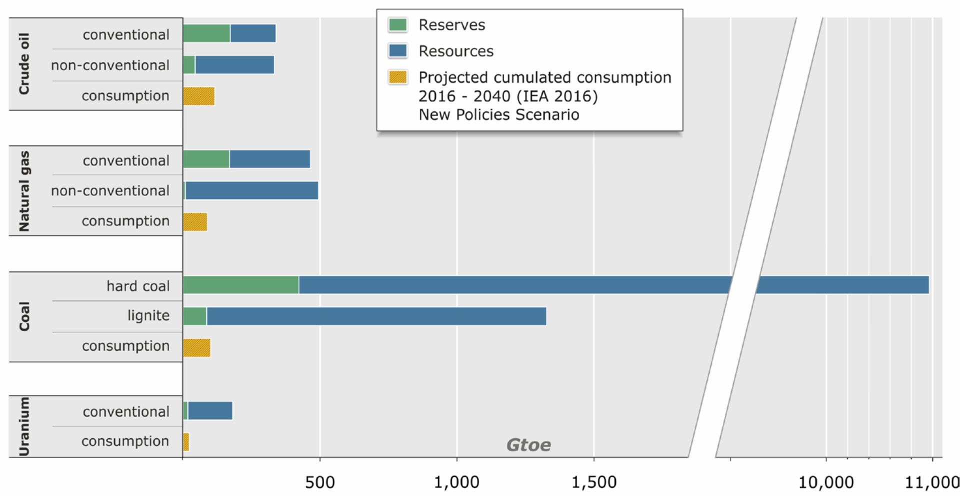

Figure 2.1: Supply and consumption of non-renewable energy resources; reprinted with permission from and based on BGR (2016)

2.2.2 Competitiveness

As competitiveness is closely linked to generation costs, the levelized cost of energy (LCOE), i.e. the discounted lifetime fixed and variable costs per unit of electricity produced, are considered in the first instance. Estimates on the LCOE for electricity generation technologies vary widely (DECC 2013, European Commission 2008b, IPCC 2014a, Schröder et al. 2013, U.S. Energy Information Administration 2015), mainly depending on assumptions regarding:

- investments, fixed and variable operating costs;
- financing conditions, described by the weighted average cost of capital (WACC);
- power plant lifetime, conversion efficiency, and utilization rate (full load hours);
- CO_2 emissions and related costs;
- discount rate.

As of today, fossil-based electricity generation still qualifies as a relatively cheap option compared to nuclear or renewable energy-based generation, albeit with a strong dependency on full load hours (Figure 2.2) and also on assumptions regarding future technology and cost developments ("learning curves").

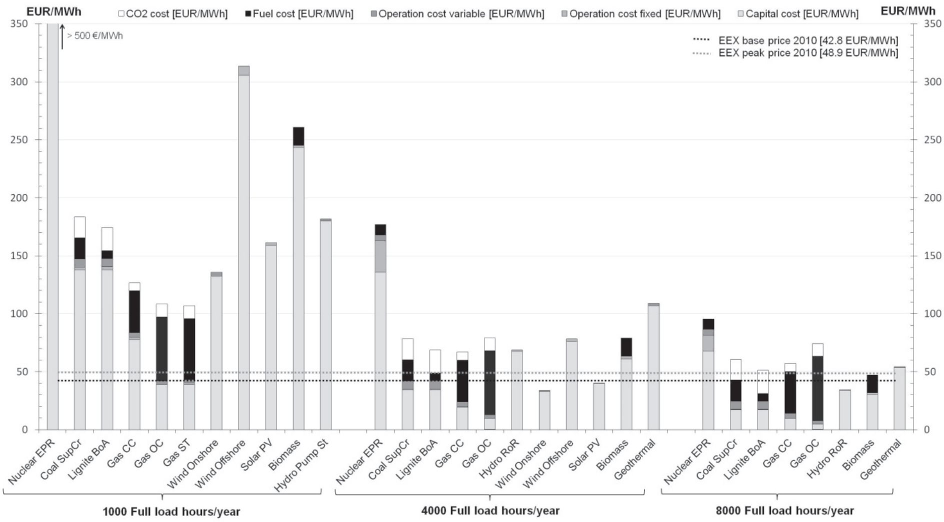

Figure 2.2: Levelized cost of electricity of different electricity generation technologies as a function of full load hours per year; reprinted with permission from and based on Schröder et al. (2013)

At the same time, using the LCOE is subject to criticism, especially when comparing intermittent renewable technologies with dispatchable conventional technologies. This is mainly because the LCOE neither reflects attributes such as security of supply (availability and flexibility), nor system integration costs or the market value of the electricity generated (Edenhofer et al. 2013, IEA 2014, Joskow 2011, Ueckerdt et al. 2013). When accounting for these factors, the relative advantage of fossil fuel-fired generation compared to intermittent renewable energies tends to increase. On the contrary, when considering environmental impacts, most of which are not reflected in the LCOE, the competitiveness of fossil fuel-fired generation is expected to decrease, as discussed in the next section.

Subsidies on extraction and use of fossil fuels

Although no immediate strength but rather representing another type of market distortion, public subsidies for fossil fuel extraction (e.g. for hard-coal mining in Germany) or consumption (e.g. tax reliefs for specific types of fossil fuel used in certain industries) contribute to the competitiveness of fossil fuel-fired power generation in many countries worldwide (OECD 2013).

2.3 Supply and demand, structural changes and a tendency of increasing flexibility

One cause for a deteriorating business context of fossil fuel-fired power generation in past years is a tendency of falling electricity demand, particularly in the industry sector. The dependency of electricity demand on macro-economic factors can be observed when looking at the economic crisis of the years 2008 and 2009, initially sparked by turbulences in financial markets. In a number of EU countries, the crisis led to a slowdown of industrial activity and hence a tendency of decreasing electricity generation (Figure 2.3).

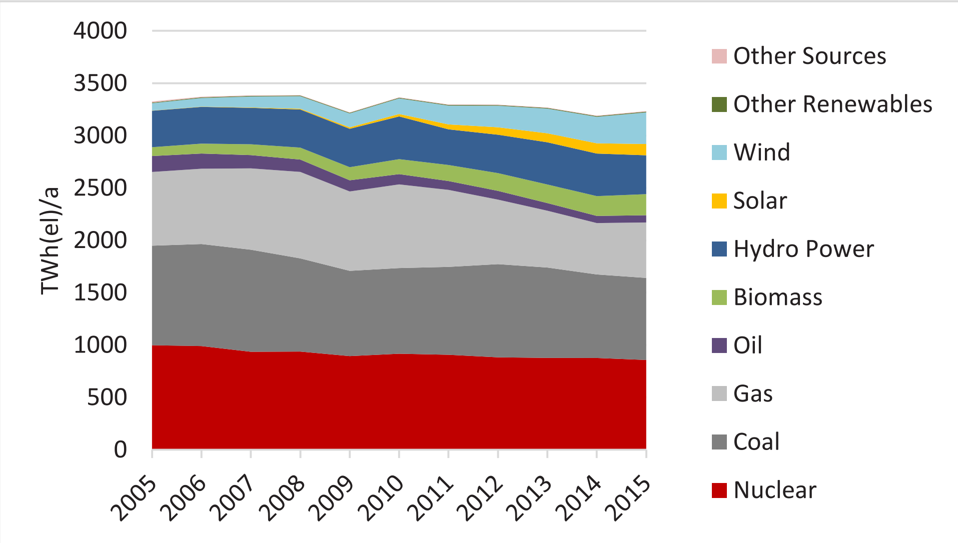

Figure 2.3: Gross EU28 electricity generation by energy carrier; adapted from eurostat (2017), using information from European Environment Agency (2015) for data aggregation purposes

Although electrification within certain sectors, especially transport and heating, could reverse this trend to some degree, more and more stringent energy efficiency measures across all sectors (Table 2.1) are expected to sustain a downward pressure on electricity demand in the future.

Another cause for the difficulties of conventional power generation, particularly evident in Germany, are structural changes within the energy sector itself, such as the fast rising share of renewable energy sources that is set to increase further (Table 2.1). Through priority market access and fixed feed-in tariffs, they have contributed to decreasing wholesale electricity market prices ("merit-order effect"), while at the same time requiring more flexibility in the system (European Commission 2015a, Sensfuß et al. 2008).

The implications of increasing flexibility, induced by variable renewable energies, have been analysed in various studies, for instance by modelling prospective energy scenarios for the German market (Brauner et al. 2012, Nitsch et al. 2012). For fossil fuel-fired power generation, these foresee a reduction in annual full load hours[1] as well as an increasing tendency of part-load operation and more frequent load shifts, generally referred to as cycling. This in turn reduces the overall energy conversion efficiency with the consequence of increasing generation costs and with different effects on atmospheric emissions (Lew et al. 2013), further discussed in section 8.4. Moreover, increased cycling bears the risk of material wear or damage, further deteriorating the power plant's competitiveness (EPRI 2001, Kumar et al. 2012).

Notwithstanding these tendencies, as long as alternative flexibility mechanisms such as storage or demand-side solutions remain under development, an important share of electric backup capacity will continue to be provided by fossil fuel power plants. Fossil fuel-fired power generation, responsible for 42.7% of gross EU electricity generation in 2015 (Figure 2.3), thus faces a dilemma. On the one hand, it is expected to remain a stabilising element in the European power generation mix to ensure security of supply. On the other hand, its profitability is under increasing pressure by the latest market developments. Capacity markets, i.e. markets that remunerate the availability of production capacity as a supplement to electricity generated, are considered as a possible solution to this dilemma and are currently being introduced in some countries, e.g. France, while other countries like Germany stick to the so-called energy only market, i.e. without dedicated capacity markets. Yet, Germany foresees alternative measures to secure supply, e.g. a strategic reserve of fossil fuel power plants (European Commission 2015a).

[1] i.e. operating hours at full nominal power plant capacity (or equivalents hereof)

2.4 Atmospheric emissions and environmental impacts

A less obvious, though increasingly important influencing factor on the competitiveness of fossil fuel power plants is their environmental performance and related regulatory constraints. Emissions into air, water or soil arise at different stages of the fuel cycle of large combustion plants, as illustrated in Table 2.2. These substances, either directly or after reacting with other chemical substances, are potential sources for environmental and human health impacts (cf. section 2.4.4).

Table 2.2: Potential emissions related to fuel cycle stages enabling electricity generation in large combustion plants; adapted from European Commission (2006b)

Source release: Air (A) Water (W) Land (L)	Substances												
	Particulate matter (PM)	Oxides of sulphur (SO_x)	Oxides of nitrogen (NO_x)	Oxides of carbon	Organic compounds	Acids/alkalis/salts	Hydrogen chloride/fluoride	Volatile organic compounds	Metals and their salts	Chlorine (as hypochlorite)	Mercury and/or cadmium	PAHs	Dioxins
Fuel storage and handling	A				W			A					
Water treatment	W								W		W		
Exhaust gas	A	A	A	A	A		A	A	A		A	A	A
Exhaust gas treatment	W				W				WL		W		
Site drainage incl. rainwater	W				W								
Waste water treatment	W				W	W							
Cooling water blowdown	W				W				W	W	W		
Cooling tower exhaust								A					

Among these impacts, the most relevant in view of policy constraints are related to emissions of greenhouse gases, e. g. carbon dioxide (CO_2) or methane (CH_4) and so-called classical air pollutants, including nitrogen oxides (NO_x), sulphur dioxide (SO_2), particulate matter (PM), ammonia (NH_3), and non-methane volatile organic compounds (NMVOC).

2.4.1 Greenhouse gas emissions: contribution of fossil fuel power plants and regulatory framework

Even though the methods developed in this thesis concern classical air pollutants, some statistical and legislative background on greenhouse gas emissions is provided due to their relevance for climate policies. Over the period from 2005 to 2014, "fuel combustion in energy industries" was causing over a quarter of reported anthropogenic greenhouse gas emission in the EU 28 (Figure 2.4). Within this category, the bulk of emissions is due to the combustion of fossil fuels, next to a minor contribution from biomass combustion.

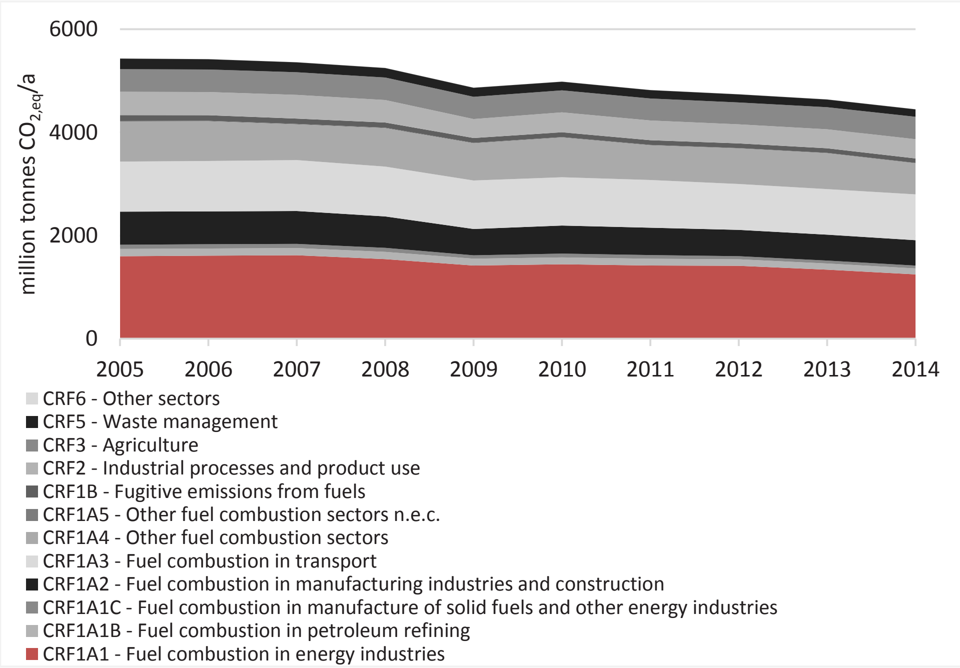

Figure 2.4: Annual EU28 greenhouse gas emissions per sector; adapted from eurostat (2016a)

Anthropogenic greenhouse gas emissions contribute to global warming, implying largely unknown, yet potentially drastic societal and economic consequences (IPCC 2014b). Greenhouse gas emissions from electricity generation and other sectors are therefore regulated under the EU's Emission Trading Scheme (ETS; Directive 2003/87/EC) that established an EU-wide market for emission allowances, whilst limiting the overall amount of emissions from the concerned sectors. Greenhouse gas emissions from non ETS sectors are regulated under the EU's so-called Effort Sharing Decision (Decision No 406/2009/EC).

Ambitious future greenhouse gas emission reduction targets set out in the European energy policy framework (Table 2.1) are expected to lead to increasing prices of emission allowances[2]. This in turn shifts generation costs of fossil fuel power plants upwards, reducing their competitiveness. A possible emission mitigation measure for greenhouse gases at fossil fuel power plants is carbon capture and storage (CCS). Yet, the economic, environmental and societal perspectives of CCS remain doubtful, as more and more demonstration projects have been scaled back or cancelled (Scott et al. 2013).

2.4.2 Classical air pollutant emissions: contribution of fossil fuel power plants

In 2014, the sector "Public Electricity and Heat Production" contributed around 17%, 47% and 4% to the reported European NO_x, SO_x, and $PM_{2.5}$ (particulate matter with an aerodynamic diameter below 2.5 µm) emissions (Figure 2.5 to Figure 2.7)[3]. These substances cause adverse effects on human health and ecosystems (cf. section 2.4.4) and have therefore been regulated in the EU for several decades (cf. section 2.4.5).

[2] So far the EU ETS has not proved very effective in increasing the price of carbon emission allowances. This is notably due to an oversupply of emission allowances, caused by forecasting uncertainties amongst others.

[3] The data, reported under the CLRTAP (cf. section 2.4.3), excludes international shipping, aviation and natural emissions (European Environment Agency 2014b). Note that there are slight differences in the sector classification as compared to Figure 2.4, however concerning only the category non-road transport.

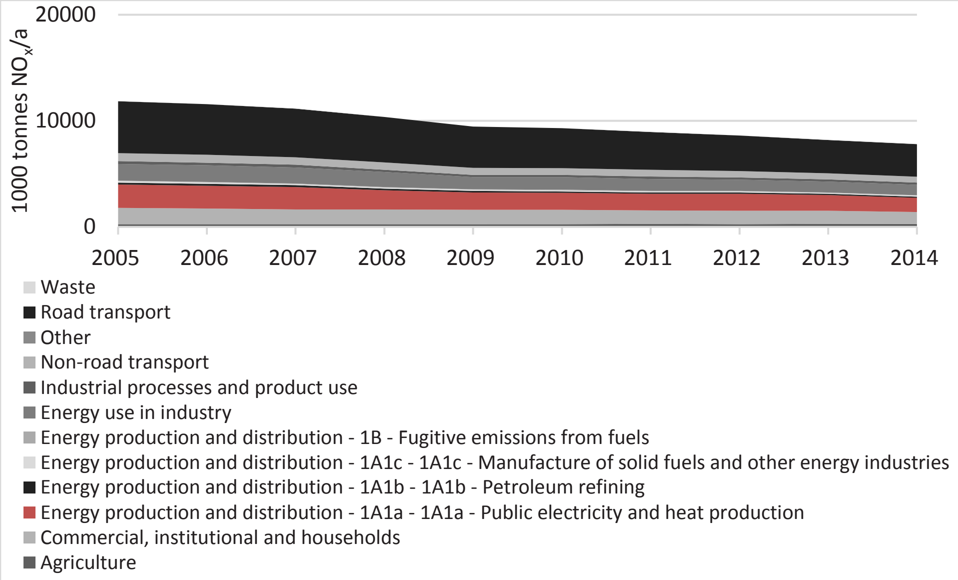

Figure 2.5: Annual EU28 NO_x emissions derived from Nomenclature For Reporting (NFR) 2014 sector classification; based on European Environment Agency (2016)

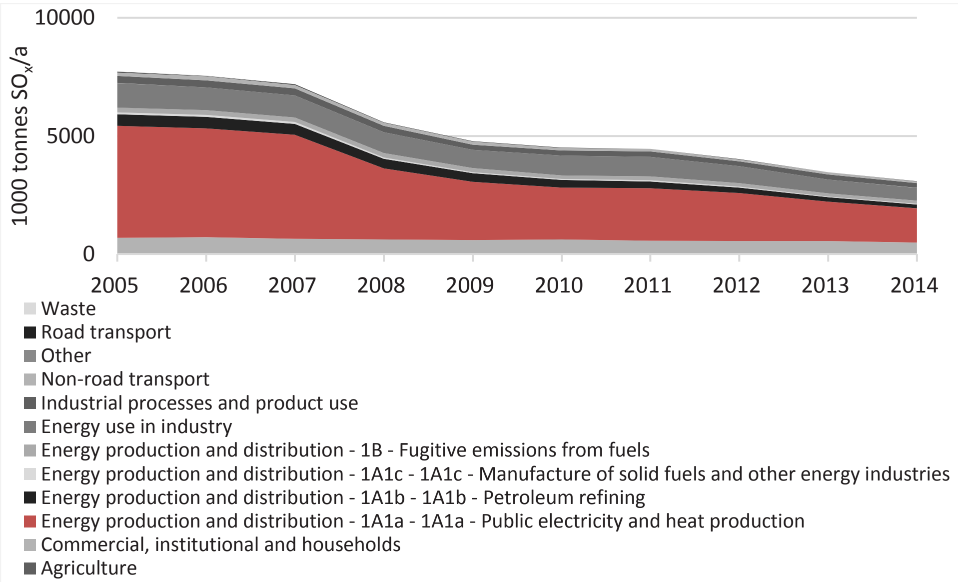

Figure 2.6: Annual EU28 SO_x emissions derived from NFR 2014 sector classification; based on European Environment Agency (2016)

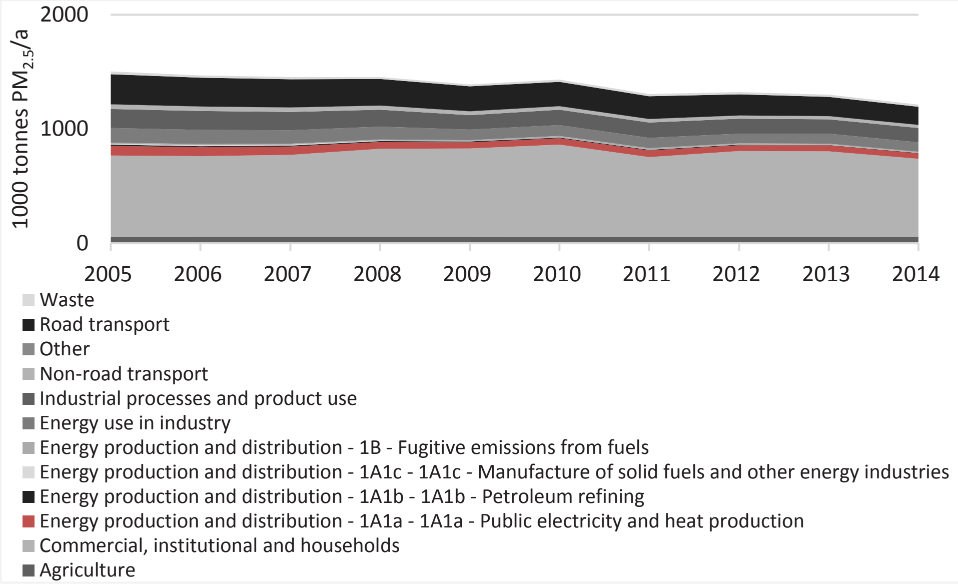

Figure 2.7: Annual EU28 $PM_{2.5}$ emissions derived from NFR 2014 sector classification; based on European Environment Agency (2016)

2.4.3 Formation mechanisms of air pollutants

Atmospheric emissions of relevance in the current thesis are formed during combustion processes. During atmospheric dispersion, they partly transform into secondary pollutants that, like certain primary pollutants, cause human health impacts (cf. section 2.4.4). Therefore, the pollutant formation mechanisms are briefly described.

Sulphur dioxide (SO_2)

Sulphur dioxide is a product of the complete combustion of sulphurous fuels such as coal or heavy fuel oil. The chemical composition and concentration of the sulphur compounds are strongly dependent on the type of fuel. For the CBA in section 7.3.1 the sulphur content in heavy fuel oil is assumed to vary between 0.22% and 0.33%.

Nitrogen oxide (NO_x)

The substances nitrogen monoxide (NO) and nitrogen dioxide (NO_2) are summarised using the generic term nitrogen oxide (NO_x). During combustion processes mainly NO is formed contained in air depends on the availability of oxygen and is favoured by high temperatures (above 1300°C). At temperatures below 1000°C the formation of NO depends on the amount of available oxygen and additionally on the nitrogen-content of the fuel (fuel NO). Prompt NO, which is less important, is formed in the oxygen-deficient part of the combustion chamber (Baumbach 1994, European Commission 2006b).

Primary and secondary particulate matter (PM)

Particulate matter is a general term referring to dust particles of different sizes. These dust particles can either be suspended in gas or in liquid. Dust particles surrounded by air are called aerosols.

A classification of PM can be performed based on the aerodynamic diameter of the particles. PM with a diameter of less than 10 µm is often referred to as fine particles. Fine particles are subdivided into:

- PM_{10}: particles with an aerodynamic diameter of less than 10 µm (including $PM_{2.5}$);
- $PM_{2.5}$: particles with an aerodynamic diameter of less than 2.5 µm.

The fraction of particles with a diameter between 2.5 µm and 10 µm is called coarse fraction. Other possibilities to characterize PM are by its chemical composition or by particle numbers.

Atmospheric PM emerges from the operation of fossil-fired power plants either as primary PM or as secondary PM. Moreover, PM also comprises natural components, such as mineral dust or sea salt.

Primary PM is formed inside the power plant and results from mineral compounds in the fuel used. Coal used in power stations, for instance, consists of 5 to 40 % mineral compounds that cannot be burned and that are transformed into different kinds of particles in conversion processes. Metal compounds can also be found as part of PM.

Secondary PM is formed where substances like SO_2 and NO_x react with ammonia (NH_3) to build so-called secondary inorganic aerosols (SIA). The products formed in these oxidizing reactions are ammonium sulphate ($(NH_4)_2SO_4$) and ammonium nitrate (NH_4NO_3). NH_3 preferentially reacts with SO_2 to form sulphate and the composition of secondary particles depends thus on the availability of both SO_2 and NO_x (Lee et al. 2015). Another type of secondary PM are secondary organic aerosols (SOA) that are formed on the basis of non- methane volatile organic compounds (NMVOCs), but whose formation mechanisms are generally less well understood than those of SIA (Hallquist et al. 2009).

Ozone

Ozone (O_3) is not directly emitted by power plants but formed in a secondary reaction. The principal substances of anthropogenic origin that are involved in this photochemical process, i.e. depending on the availability of sunlight, are NO_x, and NMVOC. They are therefore termed precursor emissions. Both the dissociation of NO_2 through sunlight as well as the oxidation of NO by NMVOC contribute to ozone formation that is therefore influenced by the availability of both substances in a non-linear way. For instance, in a typical metropolitan area setting with limited VOC and high levels of NO_x, reductions in NO_x emissions may lead to an increase in ozone formation (Finlayson-Pitts and Pitts 1993).

Carbon oxides

During combustion processes, the carbon contained in fossil fuels reacts with oxygen from the surrounding air to build CO_2. To guarantee a complete combustion, a certain excess of air or oxygen is therefore needed. As an intermediate product and being an indicator of an incomplete combustion, carbon monoxide (CO) is formed (Baumbach 1994).

Other pollutants related to combustion processes

- Non methane volatile organic compounds (NMVOC): NMVOC are emitted from fossil fuel power plants in small amounts, resulting from incomplete combustion or from fugitive emissions;

- Trace elements: Trace elements, such as arsenic (As), chromium (Cr), copper (Cu), mercury (Hg) or lead (Pb) can either be attached to particles but also, as for instance Hg, appear in the gaseous phase as a part of the flue gas, mainly related to coal combustion (European Commission 2006b). These trace elements are regularly also referred to as heavy metals even though arsenic needs to be classified as a metalloid;

- Ammonia (NH_3): Small emissions of NH_3 from can result from processes linked to the condenser of steam turbine power plants (European Commission 2006b). NH_3 is also used for NO_x reduction in the selective catalytic reduction (SCR) process. As described above, NH_3 is involved in the formation of SIA and also contributes to acidification and eutrophication.

2.4.4 Environmental and health impacts caused by classical air pollutants

Atmospheric emissions of substances like SO_2, NO_x, and PM, either directly or through the formation of secondary pollutants, have the potential to cause effects on human health, building materials, agricultural crops, and biodiversity (including ecosystem services). In Table 2.3, those environment and health impacts are displayed that are quantifiable in monetary terms using methods and models developed in the ExternE (Externalities of Energy) project series (cf. section 3.5.3) and follow-up projects. For a comprehensive list of environmental and health impacts caused by fossil fuel power plants, see Table A.1 (Appendix).

Table 2.3: Main quantifiable environment and health impacts caused by classical air pollutant emissions (European Commission 2005a, Preiss and Klotz 2008, WHO 2013a)

Impact category	Precursor substances	Effects caused by primary and secondary pollutants
Human health	PM_{10} $PM_{2.5}$ SO_2 NO_x NH_3 NMVOC	Mortality, i.e. life expectancy reduction due to short- and long-term exposure to primary and secondary PM, ozone, and NO_2 Respiratory and cardiovascular diseases due to exposure to primary and secondary PM, ozone, and NO_2 *For a detailed list of health endpoints, cf. section 6.1.3*
Building Materials	SO_2 NO_x NH_3	Damage to construction materials caused by acidifying substances (particularly SO_2)
Agriculture	SO_2 NO_x NH_3	Variation in crop yields (due to SO_2 and ozone) Acid input to agricultural soils Eutrophication of agricultural soils, avoided fertiliser use (through N deposition)
Ecosystems	SO_2 NO_x NH_3	Loss of (plant) species due to eutrophication and acidification

2.4.5 The regulatory framework targeting classical air pollutant emissions

Transboundary effects of air pollution have been addressed as early as 1979 through the United Nations Economic Commission for Europe (UNECE) Convention on Long-Range Transboundary Air Pollution (CLRTAP). Although much progress has been made since to reduce emissions from anthropogenic sources, air quality is still high on the political agenda. The EU's 7th Environmental Action Programme (EAP; Decision No 1386/2013/EU) reiterates the intention to reduce air pollution to levels "*that do not give rise to significant negative impacts on and risks to human health and the environment*".

Due to its key contribution to anthropogenic emissions of classical air pollutants, fossil fuel power plants have been regulated for many years through different types of instruments at different levels (Table 2.4).

At international level, they fall under the scope of the UNECE CLRTAP that can be classified as a common strategic objective, defining general targets that are pursued through further legislation. The CLRTAP is implemented through several sub-protocols. At EU level, these sub-protocols have been transposed through specific pieces of legislation, e.g. the National Emission Ceilings Directive (NECD; 2016/2284/EU) that ensures the respect of the Gothenburg Protocol targets. Reflecting the subsidiarity principle, EU Directives need to be transposed into member state legislation within given delays. Several examples of national transpositions for the case of Germany are given in Table 2.4.

The more recent Thematic Strategy on Air Pollution (TSAP, European Commission 2005b) can be regarded as a common strategic objective at EU level. Its targets are to be achieved, inter alia, through the Ambient Air Quality Directive (AAQD; 2008/50/EC) and also the NECD. The NECD is a special case in that it sets overall national emission ceilings to be respected in given years, but without prescribing the instruments to achieve these targets. More specific regulations are therefore required in parallel, categorised into source-oriented legislation, fuel quality and air quality standards in Table 2.4.

Among the examples, emission control at fossil fuel power plants is largely addressed by the Industrial Emissions Directive (IED, Directive 2010/75/EU), effective since January 2013. For power plants with a rated thermal capacity above 50 MW, it sets out emission limit values for NO_x, SO_2, and dust (particulate matter), unless derogations are used (cf. section 2.4.6). Emissions from power plants between 1 and 50 MW thermal capacity are covered by the Medium Combustion Plants Directive (MCPD; Directive EU/2015/2193).

Focusing predominantly on local air quality, the Environmental Impact Assessment Directive (EIAD; 2014/52/EU) sets out requirements to be respected before an operation permit can be obtained. For instance, power plant emissions shall not lead to exceedances of predefined local concentration thresholds. This condition is typically verified using health risk assessment (cf. section 3.4.2).

Table 2.4: Regulations targeting classical air pollutants and relevant to fossil fuel-fired power generation, differentiated by category and scope (own compilation)

	International	European Union	Member State level (Germany)	Local level
Common strategic objectives	UNECE CLRTAP, featuring 8 sub-protocols, targeting classical air pollutants (e.g. through the Gothenburg Protocol), persistent organic pollutants, heavy metals, etc.	Thematic Strategy on Air Pollution (TSAP; COM(2005) 446 final)		
National emission ceilings	The 1999 Gothenburg Protocol (to Abate Acidification, Eutrophication, and Ground-level Ozone)	National Emission Ceilings Directive (NECD; 2016/2284/EU)	e.g. 39. BImSchV	
Source-oriented legislation		Industrial Emissions Directive (IED; 2010/75/EU)	e.g. 4. BImSchV, 13. BImSchV	
		Medium Combustion Plants Directive (EU/2015/2193)		
Fuel quality standards		Sulphur Content of Certain Liquid Fuels (1999/32/EC)	e.g. 10. BImSchV	
Air quality standards		Ambient Air Quality Directive (2008/50/EC)	e.g. 39. BImSchV	Air quality plans
		Environmental Impact Assessment Directive (EIAD; 2014/52/EU)	Gesetz über die Umweltverträglichkeitsprüfung	

2.4.6 Best available techniques, the industrial emissions directive and the concept of disproportionate costs

In the EU, emission abatement (beyond greenhouse gases) is largely achieved by requiring that industrial installations are equipped with best available techniques (BAT), a concept introduced in the frame of the Integrated Pollution Prevention and Control Directive (IPPC, Directive 2008/1/EC, cf. European Parliament and Council of the European Union (2008)). The IPPC Directive and six other Directives were replaced by the Industrial Emissions Directive (IED, Directive 2010/75/EU, cf. (European Parliament and Council of the European Union 2010)) effective since 2013. In its annexes, the IED sets out emission limit values (ELVs) for substance releases into air and water. It reinforces the use of BAT, defined as follows (IED, art. 3-10):

"*'best available techniques' means the most effective and advanced stage in the development of activities and their methods of operation which indicates the practical suitability of particular techniques for providing the basis for emission limit values and other permit conditions designed to prevent and, where that is not practicable, to reduce emissions and the impact on the environment as a whole […].*

'available techniques' means […] under economically and technically viable conditions, taking into consideration the costs and advantages, […]"

The IED (art. 14-3) demands that BAT conclusions[4] are used as future basis for granting operation permits of concerned installations. These BAT conclusions are defined in the updated BAT REFerence (BREF) document on large combustion plants that has been adopted in 2017 and that will become effective in the year 2021. For power plant operators this implies that more stringent ELVs than those currently given in the IED's annex will become applicable in future years. In turn, power plant operators would need to decide whether employing additional emission control measures makes economic sense.

However, beyond other flexibilities, the IED (article 15-4) foresees a possibility for derogation, "*where an assessment shows that the achievement of emission levels associated with the best available techniques as described in BAT conclusions would lead to disproportionately higher costs compared to the environmental benefits*".

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[4] BAT conclusions are synthetic descriptions of Best Available Techniques (BAT) including the most relevant information, such as abatement levels of different emission reduction techniques. In addition, they will serve as reference for setting the permit conditions of future and refurbished power plants.

The following criteria may apply for costs being considered disproportionate:

"(a) the geographical location or the local environmental conditions of the installation concerned; or (b) the technical characteristics of the installation concerned." (ibid.)

The notion of "disproportionality" is crucial in this context, as it allows considering private costs and environmental benefits jointly, which refers to social CBA, further described in section 3.2. Under the increasingly tense economic conditions for power plant operators described above, the IED thus allows assessing investments in emission reduction measures on a case by case basis in terms of their overall societal efficiency. Mainly for this reason, the methodological developments in the current thesis bear an interest for power plant operators.

2.5 Summary of chapter 2

Fossil fuel-fired power generation, as a traditional and transitional energy source, continues to play a central role in the European energy mix, especially with regard to security of supply. Yet, one of its historic strengths, i.e. competitiveness, has come under increasing pressure through changes in the electricity system ("merit-order effect") and associated flexibility requirements.

Another challenge for fossil fuel-fired power generation are environmental regulations that aim at reducing the environmental and health damages related to atmospheric emissions. Source-oriented regulations, such as the EU's Industrial Emissions Directive, prescribe emission limit values, potentially requiring to retrofit fossil fuel power plants with new emission control measures. Given the increasingly challenging business context, it is crucial for operators to verify that such investments do not bear disproportionate costs. One method to do so, also in view of the requirements of the EU's Industrial Emissions Directive, is social cost-benefit analysis, introduced in the next chapter.

3 Methods: economic assessment of emission control measures

Emission control measures can be evaluated on a techno-economic or environmental-economic basis (Figure 3.1). The focus henceforth is on environmental-economic assessment methods.

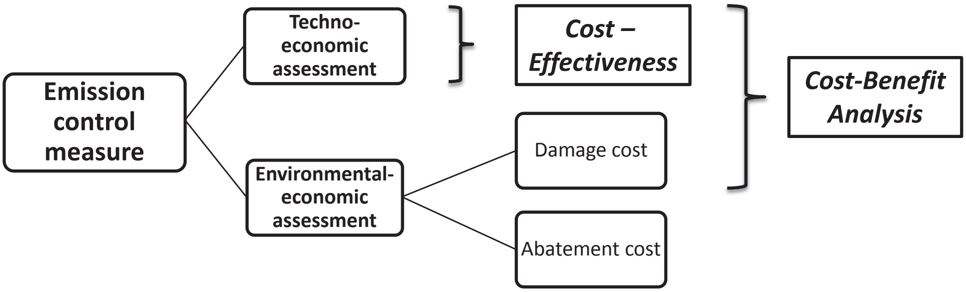


Figure 3.1: Economic methods for assessing emission control measures; derived from European Commission (2006a)

This chapter therefore starts by introducing the discipline of environmental economics, in particular the neoclassical approach to dealing with environmental problems, involving the notions of externalities and cost-benefit analysis (CBA; cf. sections 3.1 and 3.2). To seek an “optimal” pollution level, not only private cost characterisation methods are needed (cf. section 3.3), but also methods to assess the societal benefits of pollution control. The latter can be achieved using the damage cost approach (cf. section 3.4). In a state-of-the-art review (cf. section 3.5) it is shown how the presented methods have been put into scientific practice. Essentially, this review serves to identify the gaps and limitations of currently available methods that this thesis aims to overcome.

3.1 Environmental economic theory underlying the assessment of emission control measures

3.1.1 Defining environmental economics

Different definitions of environmental economics coexist (Walz 2009). Sticking closely to the wording, a straightforward definition is:

1) The application of economic theory with regard to environmental problems.

The neoclassical economic foundations underlying this definition are described in the following (cf. section 3.1.2).

An alternative definition of environmental economics, reflecting a more empirical perspective and corresponding with the methodological developments carried out in this thesis, is:

2) The assessment of environmental impacts using economic methods.

Even though the methods presented in sections 3.2 to 3.4 and applied throughout chapters 4 to 7 extend beyond the discipline of environmental economics, they serve to answer environmental-economic-related questions. Finally, environmental economics can be understood as:

3) The analysis and application of economic policy instruments with regard to environmental problems.

Although environmental policy and its impact on industrial activity are a driving factor behind the methodological developments of this thesis (cf. section 2.4.6), policy instruments themselves are only briefly mentioned in section 3.1.6. For a microeconomic-oriented discussion on environmental policy instruments, see for instance von Böventer (1995).

3.1.2 The neoclassical approach to dealing with environmental problems

Neoclassical economics is based on two central interrelated principles (Endres and Fraser 2011):

- Scarcity: Scarcity is a central pillar of microeconomics. Scarcity of environmental resources, in their role as a production factor but also for providing supporting or cultural services[5], has stimulated the development of environmental economics as a scientific discipline. In a theoretic setting, scarcity drives individuals to consider utility gains or losses of their economic (trans)actions, triggering welfare optimisation and thereby enabling the functioning of markets as allocation mechanism;

- An efficient outcome of the market mechanism, i.e. a free market functioning without the need for government intervention.

In the particular case of environmental resources however, the market mechanism is regularly disturbed due to:

- A lack of clearly defined property rights: Environmental resources often constitute public goods, characterised by non-rivalry and non-excludability in their consumption (von Böventer 1995, Walz 2009) . This makes them prone to overconsumption and partly explains another critical factor, i.e.

- A lack of market prices for environmental resources and the services they are offering to society.

As a result, environmental resources tend to be overused or degraded, implying a socially inefficient outcome of the market mechanism ("market failure"), and providing the rationale for policy intervention.

[5] Supporting and cultural services refers to the concept of ecosystem services, i.e. the direct and indirect contributions of ecosystems to human well-being (TEEB 2010).

3.1.3 Rationale for employing environmental-economic methods in the context of industrial production activities

In the context of industrial production activities, environmental resources can be characterised as (Steven 1994):

- an input factor to the production process, e.g. as raw materials;
- an absorptive capacity for outputs from the production process, e.g. environmental media like air, water or soil that absorb releases arising from industrial activities.

In the first case, not using the environmental resource as an input factor implies an opportunity cost in the sense of foregone welfare (Walz 2009). In the second case, the unrestricted use of the environmental resource implies welfare losses through health or environmental impacts caused by released substances. If these impacts of economic activity on third parties remain unpriced, i.e. outside the market, externalities arise, as further described in sections 3.1.5 and 3.1.6. Taking a public welfare perspective, environmental-economic methods are therefore instrumental to:

- put a price tag on the environment and its resources in order to enable their integration into the market mechanism;
- assist in defining the appropriate level of policy intervention, i.e. seeking a balance between all quantifiable societal (private as well as external) costs and benefits related to an industrial activity. To this end, social CBA is an appropriate method, cf. section 3.2.

3.1.4 Limitations of the neoclassical approach

Some general limitations of the neoclassical approach to solving environmental problems shall be discussed, whilst a more specific discussion in light of the results of this thesis can be found in section 8.4.

With regard to sustainable development, it is criticised that the neoclassical approach relates to the principle of so-called weak sustainability (Hohmeyer and Ottinger 1994, Neumayer 2010, Walz 2009). This concept dates back to the economist Solow, who worked on the substitutability between manmade human capital and natural capital (Solow 1974). Weak sustainability assumes a strong substitutability of these two forms of capital. It follows that, from an inter-generational perspective, it would be sufficient to maintain the total capital stock and not necessarily the natural capital as such. Apart from risking to extinguish natural capital and thereby losing critical life supporting functions, such a principle bears the risk of neglecting so-called tipping points or maximum environmental capacities, e.g. thresholds beyond which uncontrollable or irreversible damage occurs.

The assessment of environmental externalities and their use in social CBA is subject to several ethical limitations. Basically, these approaches take a utilitarian, anthropocentric viewpoint, i.e. defining well-being on the basis of human utility. It is criticised that, in many cases, this results in ignoring or underestimating the value of natural systems, for instance due to the fact that existence values and other intangible components are often difficult to consider (Ott 2014, Walz 2009). Further on, the neoclassical approach largely ignores distributional aspects across populations but also generations (Hohmeyer and Ottinger 1994). Those affected by environmental or health damage are often treated as a homogenous mass, whereas it could be worthwhile to analyse the people affected in further detail, e.g. in order to protect particularly vulnerable subpopulations. When transferring methods or parameters across space or time, methodological choices, e.g. on the discount rate to be used or on equity weighting, have implications that are often not sufficiently accounted for in decision-making. Moreover, according to Walz (2009), indirect impacts, occurring over longer time periods, are often not sufficiently accounted for when assessing environmental externalities ("monetary underestimation").

3.1.5 Definitions related to externalities theory

In economic terms, externalities are a market failure that leads to a divergence between equilibrium and social optimum, as exemplified below.

Just et al. (2004) define an **externality** "*as the case where an action of one economic agent affects the utility or production possibilities of another in a way that is not reflected in the marketplace*."

External costs are a monetary measure of negative externalities.

A notion often encountered for describing negative environmental externalities and related to the main method for measuring these (cf. section 3.4) is **damage costs**. In contrast to external costs, which, by definition, only refer to the non-internalised fraction of the total environmental damage costs (Rabl and Holland 2008), damage costs include internalised as well as non-internalised effects. Given that it is generally difficult to define the degree of internalisation of specific external effects, the term damage costs is used henceforth to describe the quantified monetised impacts due to air pollution.

Micro economic theory is primarily interested in defining **marginal effects**, meaning those effects attributable to the last (= marginal) unit of a good or service. In the given context, for instance, air pollution-related damage or abatement costs can be estimated per tonne of a substance emitted or removed, yielding marginal damage or abatement costs respectively.

3.1.6 Production externalities and internalization through policy instruments

Externalities theory is rooted in micro-economics and focuses on interactions between economic agents and related welfare losses (von Böventer 1995). Externalities linked to industrial production processes were introduced in the early 20th century by economist Arthur Pigou. After observing that the production of certain goods had unintended side-effects that are not accounted for in its market price, he distinguished private and external effects of production (Pigou 1912).

The influence of a negative production externality on welfare is illustrated in Figure 3.2, showing an equilibrium allocation of goods, described by the intersection of the supply and demand curve, in the absence (left) or presence of an externality (right).

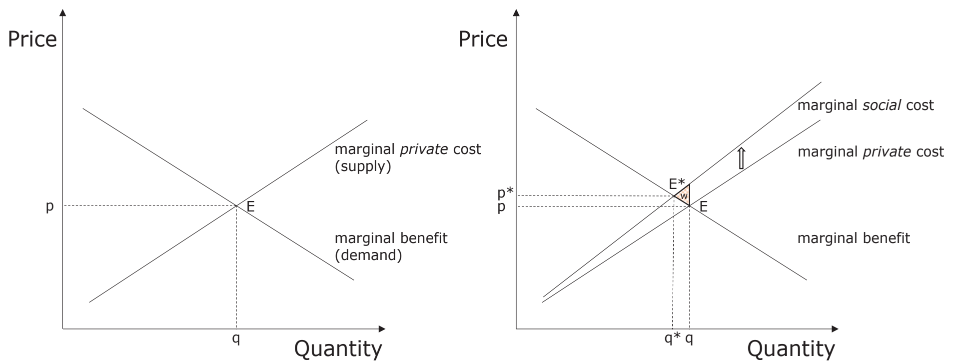


Figure 3.2: Illustration of a negative production externality; left: quantity (q) and price (p) under a competitive equilibrium (E); right: new equilibrium (E*) when internalising a negative externality, also showing associated welfare losses (w) if internalisation did not take place; derived from Pigou (1932)

When accounting for the externality, the cost curve is shifted upwards, and marginal private costs become marginal social costs. This leads to a new, socially optimal, equilibrium in which the welfare loss, originally induced by the externality, has been internalised. In this example, internalisation of a negative environmental production externality induces a price increase, whilst at the same time reducing overall societal costs by reducing the demand (from q to q*).

In addition to introducing the concept of production externalities, Pigou is also known for the so-called Pigouvian tax, a policy instrument intended to correct for negative externalities (Pigou 1932). Yet, the need for government intervention was not unanimously accepted throughout economic history. Its most prominent challenger was Coase (1960), who is known for the following theorem. It states that, where property rights are clearly defined and transaction costs are low, markets will automatically arrive at optimal allocations of resources, without the need for (government) intervention beyond potentially establishing the property rights. Effectively this process would result in an automatic internalisation of externalities.

As already mentioned, clearly defined property rights for environmental resources are often lacking. In addition, in cases of environmental pollution, transaction costs tend to be high due to the large number of potentially impacted stakeholders. For these reasons, the need for policy intervention through environmental regulations (cf. section 2.4.5) is nowadays well accepted. Two broad groups of policy instruments are distinguished, their respective advantages and shortcomings being the subject of a lively and enduring debate among economists (Pearce 2002):

- Incentive-based, also called market-based instruments. These include pollution taxes, e.g. a carbon tax, as well as marketable permits, e.g. the EU ETS;
- Quality standards, also called command-and-control measures. These include technological or performance standards, e.g. defining emission limit values as in the case of the EU IED (cf. section 2.4.3).

3.2 Cost-benefit analysis of emission control measures

3.2.1 The socially "optimal" level of pollution

Social CBA is a method to approximate the socially optimal level of pollution, at which the marginal private costs of pollution abatement equal the associated marginal social benefits (Figure 3.3).

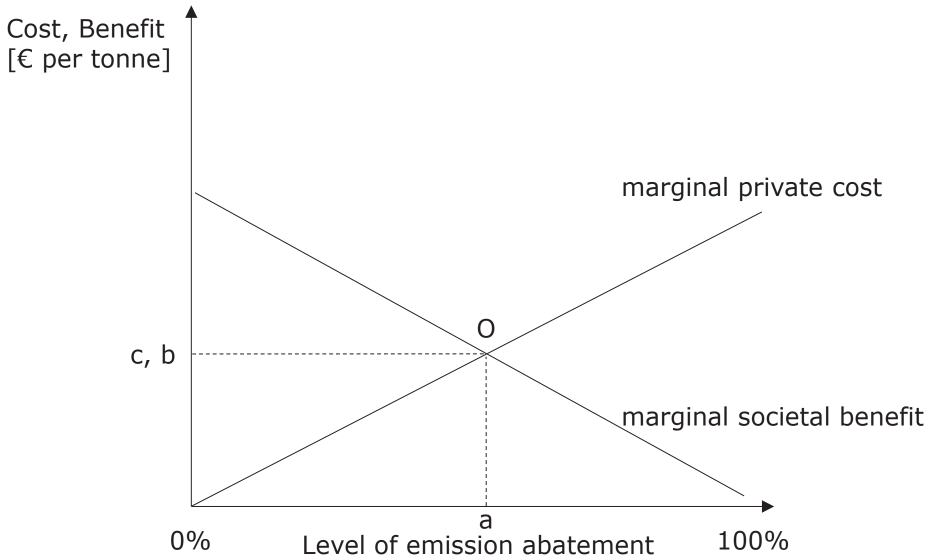


Figure 3.3: Illustration of the optimal level of pollution (O), associated marginal costs (c), marginal benefits (b) and level of emission abatement (a); derived from Samuelson and Nordhaus (2007)

The illustration represents a simplification in that the exact shape of both marginal curves depends on a variety of factors, e.g. the geographic location, the substance emitted, background conditions as well as the information available for quantifying costs and benefits. In economic textbooks, both marginal curves are typically assumed to be linear, derived from convex underlying cost and benefit curves. Yet, non-linear representations of the marginal curves also exist (Pappin et al. 2015). When regarding a bundle of pollutants at an aggregated level, it is plausible to assume a decreasing marginal societal benefit from pollution abatement, e.g. due to concentration thresholds for pollutants like ozone or nitrogen dioxide, below which further impacts are less likely to occur. Likewise, the marginal private costs are assumed to increase at higher levels of emission abatement due to technological constraints and the resulting additional efforts needed to further reduce emissions. The above example shows that a socially optimal level of emission control typically consists neither in unrestricted emissions nor in a complete elimination of emissions.

3.2.2 The decision rule in social CBA

CBA relies on a comparison of social costs and benefits, i.e. including private as well as external effects. In order to separate it from a purely private CBA, the term social CBA is used. In social CBA, the discounted benefits of a policy, programme or project (in the following simply called measure) are compared to the associated discounted costs (Pearce et al. 2006). While both absolute (based on the net present value, cf. below) and relative (based on the ratio) comparisons of benefits and costs are common, the former is considered more meaningful as it conveys information on the magnitude of the result. In case of a positive net present value, a measure is said to be advantageous or acceptable from a societal perspective, as formalised by the decision rule in equation 3.1.

Net present value: $\left\{\sum_{i,t} B_{i,t} \times (1+s)^{-t} - \sum_{i,t} C_{i,t} \times (1+s)^{-t}\right\} > 0$ 3.1

where:

- *B* refers to the benefits. In the context of this thesis, these consist of avoided health damage costs (cf. sections 3.4 and 6.1);
- *C* refers to the costs. In the context of this thesis, these consist of investment and operating costs, as a function of the measure's life time (t) and operating hours per year (cf. sections 3.3 and 6.2);
- *s* refers to the discount rate that is used to estimate the present value of future costs and benefits (cf. section 3.2.3);
- *i* refers to the affected individuals. These are determined by the scope of the analysis, i.e. the geographic extent of the modelling domain (cf. section 6.1.1).

Ex-ante (before a measure is carried out) and ex-post (after a measure is carried out) CBAs are distinguished. In the regulatory context, ex-ante CBA is more commonly used.

3.2.3 Discounting in CBA

Ex-ante CBA relies on discounting future costs and benefits, requiring value trade-offs between the future and the present. In policy appraisals, discount rates larger than zero are regularly used. This implies that future costs and benefits are given less weight than today's costs or benefits (Pearce et al. 2006). Which discounting scheme (value of the discount rate, constant vs. declining) to use remains a contentious issue and depends strongly on the context. Social discount rates, used in public policy-oriented social CBA, can be determined in two ways (Boardman et al. 2006, Harrison 2010, Zhuang et al. 2007):

- using the rate of return on risk-free investments, reflecting the time value of money, e.g. 10 year USA government bonds[6];
- using the utility-oriented "Ramsey formula", composed of
    - the pure rate of time preference, and
    - the consumption growth rate (gross domestic product (GDP) per capita) multiplied by the elasticity of marginal utility of consumption.

Discount rates for use in policy impact assessments are provided by government institutions. The European Commission (2009) recommends a real social discount rate of 4% for both costs and benefits, which is adopted in this thesis (cf. section 6.3). Real discount rates, as opposed to nominal discount rates, include effects related to price level changes (inflation). When using the Ramsey formula as an orientation, a 4% social discount rate implies a long-term average consumption growth rate of about 2% (Steinbach and Staniaszek 2015). In France, a 4.5% real social discount rate is recommended by Quinet et al. (2013), which is slightly higher than the 3% real social discount rate recommended in Germany by the Umweltbundesamt (2012) for time periods up to 20 years into the future.

Using a social discount rate in the CBA is complementary to using a private interest rate (e.g. the weighted average cost of capital, cf. section 3.3) for the conversion of investment costs into annuities. Private interest rates differ from case to case, as they reflect the opportunity cost of capital and risks related to investment decisions (Klingelhöfer 2006). For this reason, the private interest rate related to emission control investments is varied in the applied part of this thesis, demonstrating the sensitivity of CBA results with regard

[6] Given the current period of relatively low interest rates (see http://www.bloomberg.com/markets/rates-bonds/government-bonds/us, last accessed: 2017-05-18), this would lead to a lower discount rate than recommended by most government agencies.

to this parameter (cf. section 7.3.3). Private interest rates are typically higher than social discount rates given the higher risk and opportunity costs of private sector investments as opposed to public policy investments. The combined use of a private interest rate and social discount rate, as in chapters 5 and 7 of this thesis, is called the two-stage discounting procedure, adopted from Kolb and Scheraga (1990).

3.3 Techno-economic assessment of emission control measures

Techno-economic assessments are commonly used in production economics, serving for instance to:

- Identifying the most cost-effective emission control measure, e.g. assessing how a given emission reduction target can be reached in the least costly way;
- Estimating the marginal costs of emission reduction (also called marginal abatement costs), e.g. expressed in € per tonne of emission reduced. This includes a private cost characterisation of emission control measures. Under certain conditions, emission abatement costs can also be used as a proxy for environmental damage costs (cf. section 3.4);
- Optimising the functioning and performance of specific emission control measures, e.g. using simulation or optimization modelling and other engineering approaches.

Basic guidance on cost-effectiveness analysis in the context of European air quality legislation is available in the "Integrated Pollution Prevention and Control Reference Document on Economics and Cross-Media Effects" (European Commission 2006a), whose revision is pending. One central recommendation is to use the annuity method, which is presented below. On a more detailed basis, Klingelhöfer (2006) describes methods for the private financial assessment of environmental protection measures whilst distinguishing different contexts defined by different environmental policy instruments.

3.3.1 Cost characterization of emission control measures using the annuity method

In this thesis, an exemplary cost characterisation of different emission control measures at a large combustion plant is carried out (cf. section 6.2.2), based on the annuity method. For this purpose, the total costs of emission control measures are decomposed into:

- Capital Expenditures (CAPEX), also called investment, including expenditures for pollution control equipment and its installation, as well as contingency;
- Operational Expenditures (OPEX), typically composed of a fixed and a variable component. The fixed component covers costs for maintenance, insurance and wages. The variable component covers
  - electricity consumption,
  - products used for operation (e.g. reagents),
  - waste or by product-related costs (or benefits for valorised residues),
  - other equipment-related costs (e.g. replaced components).

Following the principles of the annuity method (equation 3.2), capital expenditures are decomposed into equivalent annual components that implicitly account for the underlying cost of capital, defined through the interest rate and equipment lifetime:

$$C_{ann,CAPEX} = CAPEX \times \frac{(1+i)^t \times i}{(1+i)^t - 1} \qquad 3.2$$

where:

- $C_{ann,CAPEX}$ refers to the annualised capital costs;
- i refers to the interest rate (weighted average cost of capital);
- t refers to the equipment lifetime.

The right term of equation 3.2 is also known as capital recovery factor. Total annual costs are obtained by summing the annualised capital costs and annual operational costs.

3.4 Environmental-economic assessment of emission control measures

Two principal approaches for the assessment of environmental-economic impacts related to atmospheric emissions have emerged:

- (Marginal) damage cost approach: Damage costs are assessed in a bottom-up way, starting at the emission source, tracing relevant substances throughout the environment and finishing by assessing their impacts at receptor level, e.g. human health effects. To this end, a step-wise procedure called impact pathway approach is used, further detailed below. When relating results to the last (i.e. marginal) activity unit, the term marginal damage costs is used.

- (Marginal) abatement cost approach, also termed standard-price approach: As a second-best option, marginal abatement costs serve to approximate damage costs under equilibrium conditions, i.e. where the marginal costs to prevent an extra unit of emission are assumed to equal the damage costs caused by this extra unit of emission (Baumol and Oates 1988). The approach therefore seeks to define the cost of the last (i.e. marginal) measure implemented to comply with a politically set goal, assuming that it corresponds to the welfare economic optimum. A typical field of application is greenhouse gas-related external costs, where the high level of uncertainty surrounding future impacts often prevents the application of a damage cost approach (Pearce et al. 1996, UBA 2008). A draw-back of the marginal abatement cost approach is that it cannot be used in CBA of related policies (i.e. greenhouse gas-related measures in the example given above). Since benefits are assumed to at least equal costs, the conditions for passing the CBA decision rule will implicitly be fulfilled.

Beyond these rather sophisticated approaches, benefit transfer (also called value transfer), i.e. the transfer of data or values from an original study site to a policy application site, represents another option, especially in situations where original studies cannot be carried out due to lacking resources (Navrud and Ready 2007). Given the potentially large uncertainties involved, the use of benefit transfer and in particular the compatibility between study and policy site, should be well justified.

3.4.1 Estimating marginal damage costs using the impact pathway approach

Where feasible, external costs are most precisely calculated in a site- and time-dependent way following the so-called impact pathway approach (European Commission 2005a), distinguishing steps similar to the Driver-Pressure-State-Impact-Response (DPSIR) scheme (Smeets and Weterings 1999), whilst neglecting the “Response” part. The impact pathway approach is an implementation of the marginal damage cost approach, featuring the following steps when analysing impacts of atmospheric emissions (Figure 3.4):

1. Emission scenario characterisation and air quality modelling, i.e. defining emission source specifications and associated atmospheric emissions; using atmospheric dispersion and chemical conversion modelling (or other adapted techniques) to define ambient concentration or deposition changes at receptor level;

2. Exposure assessment, i.e. estimating population (or other receptors') exposure to concentration or deposition changes. Different routes of exposure exist, e.g. inhalation or ingestion;

3. Impact assessment, i.e. assessing physical impacts using baseline data and risk functions (also called exposure-response, concentration-response or dose-response functions), derived from epidemiological or toxicological studies[7];

4. Monetary valuation of impacts, i.e. transferring physical into monetary units, based on the loss of wellbeing (or utility) of concerned individuals. In some cases, this loss can be valued using market prices (e.g. for agricultural crops or treatment of illnesses). In the case of non-market goods, such as the quality of human health, valuation is based on individual preferences, either observed (revealed preferences) or directly expressed (stated preferences) using environmental-economic methods (Markandya and Ortiz 2011, Pearce et al. 2006), cf. also section 6.1.4.

[7] Epidemiological studies provide evidence on the relations between concentration levels and risk changes through observations of population cohorts under natural exposure conditions, usually over longer time periods. Toxicological studies rely on controlled experiments to analyse the effects of substances on organisms (Savitz 1988).

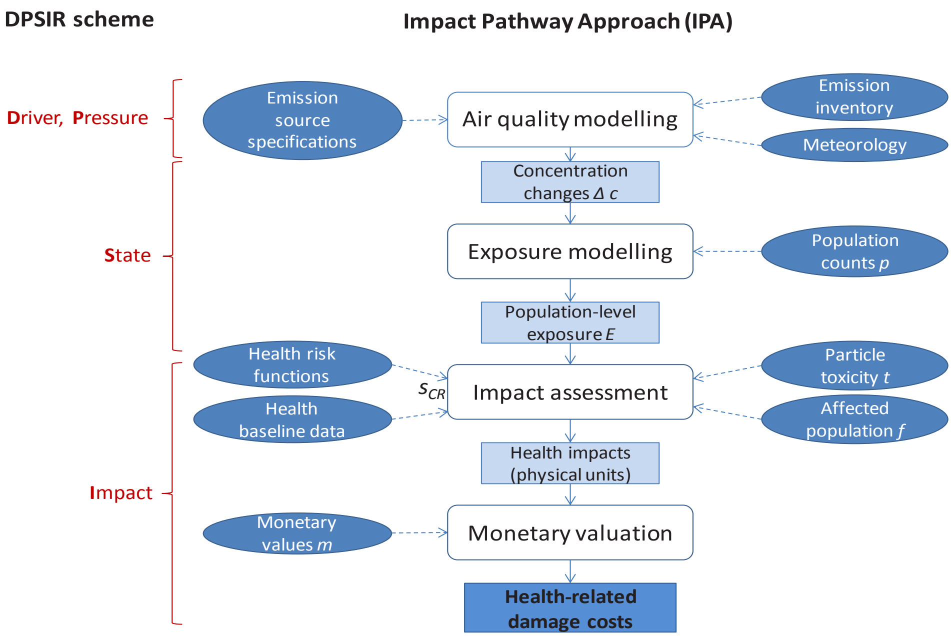


Figure 3.4: Impact pathway approach for assessing health-related damage costs following the DPSIR scheme (oval shapes = input data; rectangles = output data; rounded rectangles = assessment steps); derived from European Commission (2005a)

In principle, the impact pathway approach allows assessing the effects of emissions on various receptors, e.g. humans, agricultural crops, building materials or ecosystems. Due to their predominant weight in quantifiable damage costs (e.g. Holland 2014a), however, the current thesis concentrates on human health-related impacts. In this particular case, the last three steps of the impact pathway approach can be conceptualised by equation 3.3 that allows expressing impacts first in physical and then monetary units:

$$C_i = \sum_r \underbrace{\Delta c_r \times p_r}_{E} \times f_{i,r} \times s_{CR_i} \times \mathrm{t} \times m_i \qquad 3.3$$

where:

- C_i represents the damage costs related to health impact i, given in €$_{\text{base year}}$;
- Δc_r is the concentration change of a given pollutant in a given sub-region, expressed in $[\frac{\mu g}{Nm^3}]$ for inhalation-related exposure;

- p_r is the number of affected individuals (persons); combined with Δcr, the total population exposure E is obtained, given in $[\frac{\mu g}{Nm^3} \times person]$, calculated by summing up over all sub-regions r of the assessment domain;

- $f_{i,r}$ is the share of the population in sub-region r affected by health impact i [fraction], comprising information on the corresponding age and risk groups;

- s_{CR_i} is the slope of the impact function of health impact i, for inhalation given in $[\frac{additional\ cases}{\frac{\mu g}{Nm^3} \times person \times year}]$, merging information on the risk increase (mainly presented as relative risk and also called concentration-response function, cf. Table 6.7; here assumed to be linear with respect to concentration changes) and baseline rate of a given health impact i (IOM 2011);

- t is a factor to account for different assumptions on particle toxicity, by default and following latest recommendations (WHO 2013b) having a value of one, while values for secondary PM may deviate in sensitivity scenarios (cf. section 5.4.4.2);

- m_i is the monetary value per occurrence of health impact i, given in $[\frac{€_{\text{base year}}}{case}]$.

3.4.2 Distinguishing two policy-relevant types of health assessment

As briefly mentioned in the context of air quality-related regulations, different types of health assessment are commonly used for policy support (Table 3.1):

- Health impact assessments, often serving as an input to social CBA or for accountability studies[8]. These usually include a quantification of physical impacts at larger domains and often rely on advanced atmospheric dispersion modelling, e.g. using Euler models[9];

[8] Accountability studies aim at assessing the effectiveness of environmental or health policies, e.g. by estimating the public health benefits from respecting stricter ambient air quality standards (HEI 2003).

[9] Eulerian models rely on solving differential equations to trace the flow of fluids in the atmosphere, e.g. driven by advection, diffusion, and chemical reactions (cf. section 6.1.1.2). This makes them suitable for modelling chemically reactive substances.

- Health risk assessments, typically used in the frame of permitting procedures. These often rely on Gaussian dispersion modelling[10] and consider a limited, local spatial scale. Their main objective is to verify that toxicologically-defined risk thresholds are respected for a defined area.

Although partly relying on the same data basis, health impact assessment and health risk assessment thus differ in terms of application, scope, and results.

Following a definition of the World Health Organization (WHO 1999), "*Health Impact Assessment is a combination of procedures, methods and tools by which a policy, program or project may be judged as to its potential effects on the health of a population, and the distribution of those effects within the population.*" The term thus encompasses a variety of quantitative and qualitative methods, differing in practical details, such as considered impacts, stakeholder involvement or timing (Mindell et al. 2003). In the given context, health impact assessment, as third step of the impact pathway approach, is used as a basis for estimating damage costs. Details on the approach implemented in the current work are provided in section 6.1.3.

Environmental impact assessment, relevant in permitting procedures, encompasses a variety of physical, biological or socio-economic effects, typically relying on health risk assessment when evaluating health effects. However, the use of health impact assessment is not excluded (CDC 2015).

[10] Gaussian plume models assume that the vertical and horizontal concentration profiles of substances can be estimated using two independent normal distributions. Driven mainly by wind and atmospheric turbulence, the vertical and horizontal dispersion of a plume is calculated using simplified equations (Rabl et al. 2014). Gaussian models are therefore more suitable for modelling non-reactive substances.

Table 3.1: Comparison of the main features of health impact assessment and health risk assessment applied to emission sources; based on CDC (2015)

	Health impact assessment	Health risk assessment
Approach	Assessing the potential impacts of a measure (characterised by social, economic or environmental changes) on a population's health, including distributional effects	Assessing biophysical health risks from exposure to hazardous substances, e.g. chemicals
Application	Decision-making in voluntary and regulatory processes; using a rather broad definition of health, including physical and psychological health and general well-being	Mainly regulatory decision context, e.g. permitting of industrial facilities under the EU Environmental Impact Assessment (EIA) Directive (2014/52/EU)
Result	Physical health endpoints, e.g. cases of illnesses; potentially using aggregate metrics, such as disability adjusted life years (DALYs) or monetary valuation	Risk quotients or number of exceedances, e.g. testing whether predefined concentration thresholds or standards are respected
Scope	Local to hemispheric effects caused by one or several sources; exposure zone depending on the study design, but typically larger areas, e.g. country-level; one or several assessment years	Local effects caused by a specific source; exposure zone typically several kilometers around the source; typically several assessment years
Air quality modelling	Different kinds of atmospheric models used, depending on the study scale; regularly including atmospheric chemistry processes, e.g. the formation of secondary pollutants in Eulerian models	Typically using local-scale Gaussian modelling, regarding the dispersion of primary pollutants and with limited consideration of atmospheric chemistry

3.4.3 Uncertainty underlying the environmental-economic assessment of health damage costs

Health damage costs are inherently uncertain, as their assessment involves modelling, measurements, and expert judgements. Given their use in decision-making, particularly within social CBA, it is important to identify the nature and sources of uncertainty and discuss the implications for decision-making. This section serves to present relevant terms and definitions, uncertainty sources as well as uncertainty quantification approaches related to health damage cost assessments. A quantitative and qualitative discussion of uncertainty underlying the results of this thesis can be found in section 8.2.

Terms and definitions

Distinctly defining the nature and sources of uncertainty is not straightforward given that different definitions coexist, depending on the use context. A key question is whether variability should be considered as a type of uncertainty (the viewpoint adopted in this thesis) or rather as a distinct and separate concept, e.g. in the context of chemical risk or exposure assessment (U.S. EPA 2011b, WHO 2008). While uncertainty is precisely defined in certain domains, e.g. in the evaluation of measurement data (IPCC 1996, JCGM 2008), the multidisciplinary nature of the impact pathway approach requires a broader classification of uncertainty. Here, uncertainty definitions used in health risk assessment and greenhouse gas emission monitoring and assessment are adopted (Institute of Medicine 2013, IPCC 1996, Swart et al. 2009). A classification is proposed, covering different types of uncertainty based on their nature and the main underlying sources (Table 3.2).

Table 3.2: Uncertainty classification including the nature/type of uncertainty and associated sources; adapted from (Institute of Medicine 2013, IPCC 1996, Swart et al. 2009)

Nature/type of uncertainty	Source
Aleatory (or exogenous) uncertainty: statistical variability and heterogeneity	Natural variations in the data or system being assessed
Epistemic uncertainty: model and parameter uncertainty	Lack of knowledge about the values, causal relationships or functional forms underlying models or data
Deep uncertainty	Uncertainty/ignorance about fundamental processes or assumptions underlying an assessment

Another relevant type of uncertainty in the given context is decision-rule uncertainty due to expert choices, cf. the literature cited in Bachmann (2006). Depending on the degree of knowledge available, it may be either related to epistemic uncertainty or deep uncertainty (in cases of serious disagreement).

Uncertainties underlying the impact pathway approach

When assessing health damage costs based on the impact pathway approach, variability, as well as model and parameter uncertainty are overlapping and it is not always possible to clearly separate them. The main types and sources of uncertainty at the different stages of the impact pathway approach are (cf. the discussion in section 8.2 and (Bachmann 2006, Kim et al. 2015, Mansfield et al. 2009)):

- Air quality modelling
  - Variability, e.g. temporal and spatial variability of input data according to the respective modelling resolution;
  - Model uncertainty, e.g. due to the structure of the mathematical models and the numerical schemes used for simulating complex atmospheric chemistry processes;
  - Parameter uncertainty, e.g. related to input data regarding meteorology, emission inventory, land use data etc.;
  - Decision-rule uncertainty, e.g. choices of the numerical approximations regarding grid resolution, vertical layers, and time resolution.
- Exposure assessment
  - Variability in population exposure patterns (temporally and spatially);
  - Model uncertainty related to the spatial and temporal resolution used;
  - Parameter uncertainty, e.g. related to population statistics.
- Health impact assessment
  - Model uncertainty, e.g. due to the functional form of concentration-response curves;

- Parameter uncertainty, e.g. related to the central estimate of the concentration-response function used;
- Decision-rule uncertainty, e.g. due to choosing the mortality impact assessment approach.

- Monetary valuation
  - Parameter uncertainty, e.g. related to the valuation of intangible goods;
  - Decision-rule uncertainty, e.g. related to the components (tangible versus intangible) to be considered for valuation.

As a key difference, advancements in knowledge allow to reduce epistemic uncertainty, whereas variability (aleatory uncertainty), as an inherent feature of the system being modelled, persists. Characterising variability, e.g. through information about the magnitude of the spatial or temporal variability, is useful when it comes to decision-making. Increasing the modelling resolution generally enables to better capture variability of input data. In particular it allows estimating the potential errors induced by the use of time- or space-averaged assessment approaches (cf. section 8.2).

Common approaches for uncertainty assessment of health damage costs

Uncertainty characterization methods can be broadly classified into screening methods, qualitative approaches, deterministic approaches, and probabilistic uncertainty analyses (WHO 2008).

A typical example for a qualitative uncertainty characterisation is the so-called review of unquantified biases, describing the potentially important, but unquantified influencing factors, as well as in what way they would impact the results (Holland 2014b, U.S. EPA 2012c). Such a review is carried out in section 8.2.3.

Commonly used quantitative uncertainty characterisation methods related to the assessment of health damage costs are (Bachmann 2006, Holland 2014b, Holland et al. 2005b, Spadaro and Rabl 2008):

- Scenario or sensitivity analysis. It is used to (deterministically) assess the effect of variations in model inputs or modelling choices on variations in model outputs (cf. chapter 7 of this thesis);

- Probabilistic uncertainty assessment. A typical and widely-used example of a probability parameter uncertainty assessment is Monte Carlo analysis. Essentially, it consists of two steps. First, probability distributions around central value estimates of key parameters used for health damage cost assessment are defined; second, a sufficiently large number of modelling runs is conducted, during which input parameters are simultaneously and randomly varied according to their predefined characteristics. The results give a broader overview on uncertainties than a single sensitivity analysis and also permit to identify the most influential input parameters. Yet, the quality of the results strongly depends on the quality of the input data, as also discussed in section 8.2.

3.5 State-of-the-art review including gap analysis: economic assessment of emission control measures

This section aims to review existing approaches for the techno-economic, environmental-economic, and CBA-based assessment of emission control measures (Table 3.3). The overview is not intended to be exhaustive, but includes those publications considered most relevant for the context of this thesis. Specific models for an environmental-economic damage cost assessment are presented in section 3.5.7. The following filter criteria were used in order to obtain relevant results from scientific databases and related literature:

- assessment of emission control measures for classical air pollutants released from point emission sources in the energy sector in a Western European (or North American) setting;

- applied research, i.e. involving some kind of modelling and results;

- research with a strong link to environmental policy-making.

Moreover, guidelines for deriving private costs of emission control measures were considered.

Table 3.3: Overview on exemplary studies assessing emission control measures of classical air pollutants in the energy sector, including information on their scope and models used (own compilation)

	Aggregated level (e.g. sector or region)	**Site-level**	**Site-level and accounting for variability**
Techno-economic assess-ment	**EU** Economic assessment of BAT (Schultmann et al. 2001) Reference installation approach (Geldermann and Rentz 2004) BREF large combustion plants (European Commission 2006b) EU-wide emission control strate-gies, GAINS model (Amann et al. 2014) **National level** Sector-specific emission control scenarios, Germany (Breun et al. 2012) Sector-specific emission control scenarios, France (MEEM 2016)	**EU** Industrial emission control measures at site level (Rentz 1979) **USA** PM control strategies, US context (Atkinson and Lewis 1974)	**EU** Site-specific emission control costs (TFTEI 2015) **USA** Site-specific emission control costs (U.S. EPA 2002)
Environ-mental-economic assess-ment	**EU** Electricity sector damage costs in EU countries, ExternE project se-ries, (European Commission 1999b, 2004) EU and German damage cost as-sessment, EcoSense model, e.g. (Krewitt et al. 1999) Sectorial damage costs in four EU countries (Droste-Franke et al. 2005) EU and Danish damage cost assess-ment, EVA model, cf. section 3.5.7 (Brandt et al. 2013) **National level** Social costs of energy consumption, Germany (Hohmeyer 1988) Sectorial damage costs in Germany (Schwermer et al. 2014), (van der Kamp et al. forthcoming)	**EU** Damage costs EU, Risk-Poll/Uniform World Model, cf. section 3.5.7 (Curtiss and Rabl 1996, Rabl and Spadaro 2000) Damage costs Poland, EcoSenseWeb model, cf. section 3.5.7 (Czarnowska and Frangopoulos 2012) Damage costs of European industrial facilities (European Environment Agency 2011, 2014a) **USA** US Damage cost assess-ment (Mendelsohn 1980, Muller and Mendelsohn 2007)	**EU** Damage costs of Dan-ish facilities, EVA model, cf. section 3.5.7 (Andersen et al. 2006) Health damage costs of power plants with variable operation profiles in Europe, cf. section 6.1

	Aggregated level (e.g. sector or region)	Site-level	Site-level and accounting for variability
Social cost-benefit analysis	**EU** EU air quality policy (Holland 2014a, Watkiss et al. 2005) **USA** US air quality policy (U.S. EPA 2011a, 2012c) **National level** French air quality policy (MEEM 2016)	**EU** Case study of a coal-fired power plant, cf. chapter 5 (Bachmann and van der Kamp 2014) UK IED derogation cost-benefit analysis tool (Ricardo AEA 2017)	**EU** Social CBA methodology, cf. chapter 6

3.5.1 Early pioneering work

In one of the first works identified, Atkinson and Lewis (1974) used source-receptor transfer coefficients based on Gaussian air quality modelling to respectively define the effect of reducing PM emissions in a source region on concentration changes in receptor regions in the USA. In parallel, they also estimated emission control costs, however without proceeding towards a CBA, as no health impact assessment was carried out. The study's main objective was to define cost-minimal reduction strategies that would enable regions to respect given air quality standards. As a limitation in scope, the Gaussian modelling only considered primary PM dispersion at a restricted geographic scale.

Extending the former work, Mendelsohn (1980) coupled a techno-economic assessment with an environmental model that allows incorporating environmental damage costs. However, this work also steps short of a classical CBA, mainly due to the author's lack of confidence in central monetary values that could be assigned to health impacts. Using Gaussian air quality modelling based on simplified chemistry as well as dose-response functions that were qualified as relatively uncertain, health impacts were quantified primarily in physical units. In a sensitivity analysis, the author sought to define what levels of health costs would be needed in order to break even with the abatement costs of distinct abatement measures. Almost 30 years later on the same author co-published an article in which a damage cost assessment methodology is applied in the US context (Muller and Mendelsohn 2007).

As one of the pioneers in Europe, Rentz (1979) carried out extensive work on the techno-economic assessment of industrial emission control measures. Notably, he pointed out the non-feasibility of CBA, mainly due to lacking capabilities for a regionalised, site-specific exposure assessment that accounts for atmospheric chemistry of a "pollutant mix". In addition, lacking scientific capabilities for a health impact assessment were mentioned. A notable recommendation given by Rentz (and followed in the current thesis) is to proceed towards the "disaggregated level" for an assessment of emission control measures.

Regarding the environmental-economic assessment of energy conversion technologies, Hohmeyer (1988) can be counted among the pioneers in Europe, estimating social costs of energy, including health damage costs. As a limitation, he relied on damage cost factors from literature studies, without carrying out an original modelling for the German context. Social costs of energy were used to point out the competitive disadvantage of renewable energy sources compared to conventional energy sources in the electricity market, brought about by the lacking consideration of the external costs. The study thus made an early call for renewable energy support schemes.

3.5.2 Costing methodologies

Reviewing costing methodologies is hindered by the fact that emission control costs are site- and context-dependent. For this reason, this review focuses on methodological background documents and spreadsheets that permit the evaluation of power plant-specific emission control costs. In parallel, studies describing reference installations or technologies and respective abatement costs for use at a more aggregated level are also considered.

A comprehensive guidance for defining air pollution control costs of technical measures at fossil fuel power plants is given by U.S. EPA (2002). Parts of the same methodology have also been implemented in spreadsheets by the Task Force on Techno-Economic Issues (TFTEI 2015)[11]. Although the version of the TFTEI spreadsheet that is used in the frame of the current work (cf. section 6.2) enables the user to consider part-load operation to some extent, the necessary data could not be obtained and therefore efficiency losses due to part-load operation were not considered.

[11] Formerly known as Expert Group on Techno Economic Issues (EGTEI), working under the United Nations Economic Commission for Europe (UN-ECE) on emission control costs amongst other.

On a more aggregated and technical level, the BAT REFerence (BREF) document for large combustion plants (European Commission 2006b) provides information on emission control measures, including typical ranges of abatement costs and emission levels (cf. section 2.4.6). Dedicated scientific methods supported the preparation of the BREF document, e.g. the reference installation approach (Geldermann and Rentz 2004, Nunge 2001). It seeks to define representative installations and associated abatement costs for use at a more aggregated level, e.g. to be used in integrated assessment models. Another work dealing with the economic assessment of BAT by Schultmann et al. (2001) provides a systematic approach for the assessment of investments and operating costs. It aims at providing a consistent methodological basis for the economic assessment of BAT from different sectors.

3.5.3 The Externalities of Energy project series

The most widespread methodology to estimate external costs related to atmospheric emissions in the energy sector has been developed in a joint US-European research initiative (European Commission 1995, Oak Ridge National Laboratory and Resources for the Future 1992). On the European side, this project became known as the Externalities of Energy (ExternE) project series (European Commission 1999a, 2005a). The impact pathway approach has been developed and promoted within ExternE as a bottom-up method to estimate environmental and health damage costs related to atmospheric emissions. Owing to advances in atmospheric modelling and in health impact assessment, site-specific assessments of impacts caused by energy conversion technologies were enabled, implemented in different tools (cf. section 3.5.7). The development of the health impact assessment methodology during more than 20 years is scrutinised in chapter 4. Besides assessing external costs of individual power plants in Europe, the ExternE methodology also provided the scientific basis for a more aggregated assessment of air quality scenarios at European level.

3.5.4 The GAINS model and its role in European policy-making

The Greenhouse Gas and Air Pollution Interactions and Synergies (GAINS) integrated assessment model has played an important role in air quality-related policy proposals at the European level. The model (or modelling framework, as it relies on inputs from other models), is used to determine cost-effective emission control strategies per sector and country that enable achieving predefined emission reduction targets, e.g. related to certain health objectives, at lowest costs (Amann 2012). As such it takes into account both the emission and exposure level, the latter being used to quantify health impacts.

For data input, GAINS relies on sector-level activity data from fundamental models, e.g. PRIMES for the energy sector (Capros et al. 2013), in certain cases complemented by national activity projections delivered by national governments. Besides, it incorporates emission factors, emission control measures and related costs. Its main output are optimised emission scenarios per country and sector, including reduction technique deployment, underlying private abatement costs as well as associated health benefits. For the latter, the impact pathway approach is used. In order to minimise emission control costs while assuring the respect of European-wide health targets, European Monitoring and Evaluation Programme (EMEP) source-receptor matrices[12] are used. These source-receptor matrices are derived from a 15% emission reduction per grid cell and are therefore suitable to define impacts at sectorial level, whereas individual emission sources are generally not well represented by this approach (cf. also discussion below). Source-receptor matrices with similar characteristics are underlying the EcoSenseWeb model (cf. sections 3.5.7 and 5.2), used amongst other to estimate sectoral reference damage costs for Germany (Schwermer et al. 2014, van der Kamp et al. forthcoming).

National research activities often aim at refining components used in integrated assessment models at larger scales. For the case of Germany, Breun et al. (2012) used simulation modelling to determine detailed future emissions from the industrial, residential, and traffic sectors. The underlying objective was to study the influence of policy instruments on these emission levels. In France, a concerted effort at national level was undertaken to develop an assessment methodology comparable to that of GAINS and including a CBA of sectorial emission control measures (cf. next paragraph).

3.5.5 Use of CBA for policy support and at power plant level

CBA has been extensively used for decision support in EU air quality policy-making, e.g. in the frame of the Clean Air For Europe (CAFE) Programme (Holland et al. 2005d) that led to the Thematic Strategy on Air Pollution (TSAP: European Commission 2005b) and also for the more recent EU Clean Air Policy Package, released in 2013 (European Commission 2014a, Holland 2014a). In both cases, the CBA builds upon the ExternE methodology and

[12] According to Tarrasón (2009), “source-receptor (SR) matrices give the change in various pollutant indicators in a receptor country, sub-area or grid square, resulting from a change in anthropogenic emissions from a specific emitter grid area, sub-region or country.”

on the GAINS model. At French national level, CBA results (amongst other indicators) served to analyse the impact of future sectorial emission control measures (MEEM 2016). Moreover, CBA is routinely used in the USA to estimate the impact of air quality-oriented policy proposals, e.g. related to the Clean Air Act (U.S. EPA 2011a). While the scientific assessments underlying these policy proposals aim at defining future sectorial emission reductions at an aggregated level, they are not concerned with assessing individual industrial emission sources.

3.5.6 Social CBA in the presence of variable operation profiles

Most available models for the site-specific impact assessment consider local background conditions in terms of population densities, meteorology or emission inventories. The level of underlying detail, however, differs.

For instance, the damage cost assessments by the EEA (European Environment Agency 2011, 2014a) combine site-specific emission data of large industrial facilities with national average damage cost factors per tonne of emission, partly adjusted for sector particularities, e.g. emission heights. In this and other cases, e.g. the EcoSenseWeb model, EMEP source-receptor matrices are the basis for deriving region- or country-specific damage costs. Accordingly, the precision of the results depends on the quantity of emissions and the modelling resolution, as emissions are assumed to arise homogenously within a source cell. Moreover, the precision of damage estimates depends on the emissions' time profile throughout the year. Source-receptor matrices are usually developed assuming a decrease in emissions among all sources within a given source region, e.g. by 15% in the case of the EMEP matrices (Tarrasón 2009). Due to this static nature, they are not suitable for assessing point sources with variable operation profiles.

A dedicated air quality modelling, e.g. as proposed in the current work or in the case of the EVA model (Andersen et al. 2006), allows to consider space as well as time-variable factors in the damage cost assessment and therefore qualifies to assess impacts from sources with variable operation profiles. At the same time, such a dedicated modelling is costly in terms of time and computer resources and applications remain therefore often constrained to the aggregated level so far. The latter is also true for the case for the EVA model and its available results.

No studies could be identified that provide a methodology for social CBA of emission control measures at point emission sources with variable operation profiles. The current thesis intends to fill this gap by first developing a site-specific damage cost assessment approach, as described in section 6.1. In a next step, the damage cost assessment is integrated with an economic assessment within a framework for social cost-benefit analysis. The costing methodology takes into account the operation hours per year as well as the equipment lifetime. This way, variable operation profiles are covered to a certain degree, cf. section 6.2.

3.5.7 Models for health damage cost assessment related to atmospheric emissions from point sources

While noting that a number of guidance documents and spreadsheets exist that contain (national average or sector-specific) damage costs per tonne of pollutant emitted (DEFRA 2015, European Commission 2006a, Preiss et al. 2008, Schwermer 2007, Schwermer et al. 2014, van der Kamp et al. forthcoming), the focus here is on models that allow estimating damage costs of individual point emission sources.

Table 3.4 summarises the features of selected assessment tools. These have been classified according to the following criteria:

- ability to assess individual point emission sources or multiple emission sources;
- air quality modelling capabilities;
- health impact assessment and monetary valuation in terms of up-to-dateness of parameters and geographic context for which parameters are provided;
- spatial and temporal resolution, e.g. possibility to consider variable emission patterns, the size and the resolution of the modelled zone;
- input data needs;
- access rights, i.e. stating whether the tools are freely accessible.

EcoSenseWeb, Risk Poll, and the EVA model enable a site-specific assessment of air pollution impacts in Europe, partly including the local domain with a limited spatial extension (Andersen et al. 2006, Preiss and Klotz 2008, Spadaro 2004). Some models, such as EcoSenseWeb or RiskPoll, allow assessing damage costs of additional impact categories beyond human health, e.g. crops or materials, which are not further considered here.

The overview reveals that the available tools are not adequate for the purpose of the current thesis. Among the freely accessible tools, only EcoSenseWeb and Risk Poll qualify for an application in the European context, envisaged here. A major limitation of Risk Poll is its simple air quality modelling that does not allow for a proper consideration of time and space dynamics. In addition, the input parameters for health assessment and monetary valuation included in Risk Poll are not up-to-date. Even though the feasibility of using EcoSenseWeb for a CBA of pollution abatement is demonstrated in this thesis (chapter 5), a major shortcoming regards the assessment of impacts from point sources with variable operation profiles. In the parameterised air quality model underlying EcoSenseWeb, atmospheric emissions are assumed to occur evenly throughout the year, contrary to the characteristics of a flexible operation mode. Moreover, the local assessment model in EcoSenseWeb only considers effects due to primary particles and uses a relatively coarse spatial modelling resolution, making it unsuitable to capture local effects in sufficient detail (cf. the weak point analysis in section 5.5.2). Consequently, an extended health damage cost assessment framework is developed and implemented in this thesis, briefly characterised at the bottom of Table 3.4 and more thoroughly described in section 6.1.

Table 3.4: Comparative overview on selected assessment tools to quantify human health-related damage costs from point emission sources (own compilation)

	Emission source	Air quality modelling	Health impact assessment	Monetary valuation	Spatial resolution (km grid)	Temporal resolution	Input data needs	Access rights
EcoSenseWeb 1.3 **(Preiss and Klotz 2008), cf. section 5.2**	Point source	Parameterised Eulerian model (Europe); Gaussian model (local), no specific model runs	EU average (year 2009); not directly modifiable	EU average (year 2009); not directly modifiable	Northern hemisphere (100), Europe (50), local (10)	1 year (current/future); constant emission rate	Low (cf. Table A.2)	License fee
Risk Poll 1.052, release 2012 **(Rabl et al. 2014, Spadaro 2004)**	Point source	Integrated simplified dispersion model, no specific model runs	EU average (year 2005); modifiable	EU average (year 2005); modifiable	Europe; other according to data availability	1 year; constant emission rate	Low / high (depending on precision level)	Free
EVA model **(Andersen et al. 2006, Brandt et al. 2013)**	Point/ multiple sources	Integrated Eulerian model, specific model runs possible	Adapted EU average (year 2011); modifiable	Adapted EU average (year 2011); modifiable	Flexible; Northern hemisphere (150), Europe (50), Denmark (16.7)	1 year; variable emission rate	Data integrated (high otherwise)	Restricted
BenMAP CE 1.0.8, release 2014 **(U.S. EPA 2012a)**	Point/ multiple sources	Not integrated; external monitoring or modelling data needed	US/China average (year 2014); modifiable	US/China (year 2014); modifiable	Flexible (depending on air quality modelling)	1 year; depending on air quality model	Data integrated (high otherwise)	Free
Current thesis, **cf. section 6.1**	Point/ multiple sources	Not integrated; Eulerian model; optional local (Gaussian) 'plume-in-grid' modelling	EU average/ France (year 2015); modifiable	Adapted EU average (year 2015); modifiable	Flexible; three geographic domains with different resolutions	1 year; variable emission rate	Data integrated (high otherwise)	Restricted

3.5.8 Overview: transferring a decision-making framework from the public policy to the business domain

Table 3.5 shows a proposal regarding the main methodological aspects to consider when transferring the social CBA methodology for the assessment of emission control measures from the public policy to the business domain. The implementation of these elements into a methodological framework is described throughout chapter 6. Further information related to uncertainty is provided in section 8.2. Final recommendations regarding the methodology transfer are provided in section 8.5.

Table 3.5: Proposal of elements to consider when transferring a social CBA methodology for the assessment of emission control measures from the public policy to the business domain (own compilation)

	Public policy EU/sector/country	⇨	**Business** Specific site
Objective of social CBA			
	Impact assessments related to policy-making at European or national scale		Verifying whether costs of emission control measures are disproportionate
Techno-economic cost assessment			
General approach	Reference technology approach, annuity approach		Site-specific assessment, annuity approach
Environmental-economic benefit assessment			
General approach	Damage cost assessment: unspecific average cost factors or parameterised modelling results, considering sectoral particularities such as height of release		Damage cost assessment: site-specific approach
Modelling resolution	Low to medium for aggregated assessments		High, reflecting the specific operating site and conditions

3.6 Summary of chapter 3

Externalities are the unpriced side effects of (trans)actions between economic agents, e.g. negative environmental impacts caused by industrial production processes. When expressed in monetary terms, they are called external costs or damage costs. To correct for this market failure, governments seek to internalise externalities through policy measures up to the point where marginal damages equal marginal abatement costs.

A prerequisite to support the internalisation process are techno-economic and environmental-economic assessment approaches for emission control measures. Social cost-benefit analysis combines both types of approaches, seeking to define the net present value of control measures by considering private and external effects. Private costs of emission control measures are most appropriately estimated using the annuity approach. Health damage costs related to atmospheric emissions are typically estimated using the impact pathway approach. The impact pathway approach has been implemented through various tools that account, to some degree, for the specific site of an emission source, however with limited temporal and spatial resolution. This is particularly critical for assessment at power plant level with variable operation patterns. New methodological advancements are therefore proposed in this thesis. By use of a highly time- and space-resolved modelling, the limitations of currently available assessment tools shall be overcome (cf. chapter 6). Specific case studies are used to conclude on the added value of such a dedicated modelling approach compared to alternative approaches (cf. chapter 7). Feeding into a social cost-benefit analysis framework for the assessment of emission control measures at site level, the methodology enables public administrations and industry alike to verify that scarce resources are spent efficiently to enhance overall societal welfare.

4 Health damage cost assessment: in-depth methodology review and adaptation

This chapter features an in-depth health damage cost methodology review and adaptation, covering a timespan from the beginnings of the ExternE project series to the EU's recently released 2013 Clean Air Policy Package. It serves to identify the main methodological influencing factors on health damage costs and as such it supports informed choices of assessment parameters (cf. section 6.1) and of potential levers for sensitivity analyses (cf. section 7.3.3). The contents are largely based on van der Kamp and Bachmann (2015).

4.1 Context and objectives

In the context of preparing the EU's 2013 Clean Air Policy Package (European Commission 2014a), the World Health Organization carried out a meta-analysis that led to the publication of a new set of recommended concentration-response functions (WHO 2013a) for use in cost-benefit analysis (CBA) of air pollution policies. This can be seen in a tradition of changing modelling components and assessment parameters throughout different EU projects over time, resulting in differences in published damage costs (Droste-Franke 2005, European Commission 2003, Krewitt et al. 1999, Krewitt et al. 1997, Rabl et al. 2011, Spadaro and Rabl 2002). A few studies analysed the link between methodological assumptions and quantified damage costs, covering the years 1995 to 2005 at most (Krewitt 2002, Krewitt and Schlomann 2006). Given the manifold scientific developments over the past 20 years and the continued relevance of air-pollution-induced health costs for policy-making both in and outside Europe, the aim of this chapter is to deepen these analyses on the impacts of methodological developments on the magnitude of quantified damage costs and extend these to the present. One important contribution to a better scientific understanding is to disentangle the influencing factors that are otherwise hidden in aggregated damage cost estimates. This is also of relevance with regard to identifying the most influential parameters to be used in sensitivity analyses of damage costs.

To illustrate developments in terms of exposure modelling, risk assessment and monetary valuation, a coal-fired power plant unit located in Western Europe is used as a case. Since quantitative results exist only for projects up to the year 2009, a methodological adaptation is carried out in order to also produce damage cost estimates based on newest European recommendations. This analysis focuses on the variability in damage cost estimates arising from heterogeneous methodological choices. Methodological influences are assessed independently of operational or geographical variations, addressed in chapter 5.

4.2 Approach

Four implementations of the impact pathway approach (termed IPA implementations henceforth) and a sensitivity scenario for the most recent implementation are compared: ExternE1998, New Elements for the Assessment of External Costs from Energy Technologies (NewExt2004), New Energy Externalities Developments for Sustainability (NEEDS2009) and Year2013/2013*.

These are applied to an exemplary emission point source for which damage costs had already been quantified before, i.e. a 600 $MW_{electric}$ pulverised coal combustion unit, located in Western France. To ensure comparability, technical emission source specifications were kept constant. For ExternE1998 and NewExt2004, published damage costs were used (European Commission 1999a, b, 2004). The NEEDS2009 results were estimated using the tool EcoSenseWeb (cf. section 5.2) and subsequently updated to obtain damage costs for Year2013 and Year2013* (cf. section 4.3).

While the data provided in section 4.3 helps to explain most of the observed changes in damage costs, disentangling the influence of single parameters is not straightforward. To quantitatively estimate their influence on damage costs, the following elements are successively analysed.

First and regarding population exposure (step 1), the question is: which damage costs result if the population exposure modelling of the most recent tool is used instead of the original exposure modelling? Aggregated population exposure figures per pollutant from the NEEDS2009 assessment were derived and then respectively combined with damage costs per pollution increment from the original 1998 and 2004 IPA implementations (cf. section 4.3.4).

In addition to an updated exposure modelling, the next question concerns particle toxicity (step 2): what is the impact of using the current recommendation of equal particle toxicity? To this end, results from step 1 were adapted in terms of particle toxicity. Factors applicable to secondary PM were defined by using the toxicity coefficients as stated in section 4.3.3 and assuming a mass ratio of nitrates and sulphates of 2:1, derived from dedicated EcoSenseWeb calculations.

Among all health endpoints, long-term mortality accounts for the largest share in total quantified health-related damage costs (cf. section 4.4.1). The influence of updating the related impact assessment parameters (risk function as well as monetary valuation) according to NEEDS2009 is assessed on top of the outcome of step 2 (updated exposure modelling and updated particle toxicity).

4.3 Models and data used

4.3.1 The tool EcoSenseWeb, its case study application and updates

Part of the damage costs were calculated with the web-based software tool EcoSenseWeb 1.3 (cf. section 5.2), developed in the NEEDS project (Preiss and Klotz 2008). Choosing this model allows comparison with the ExternE1998 and NewExt2004 results, calculated with its predecessor, the EcoSense desktop tool (IER 2004). EcoSenseWeb is applicable to stationary point emission sources in Europe. Data needed for a site-dependent externality assessment are provided, i.e. receptor data, impact function slopes, and monetary values. Only health impacts caused by classical air pollutants are considered here.

As a novel feature, EcoSenseWeb results are manually adapted in terms of impact functions and monetary values to reflect the latest expert recommendations at European level. Two parameter sets were defined, representing a base case and a sensitivity case with differing mortality valuation, denoted by Year2013 and Year2013*, respectively (cf. sections 4.3.4 and 4.3.5).

4.3.2 Exposure modelling

Two types of regional (European-wide) air quality models were used (Table 4.1), i.e. a Lagrangian model and a (parameterised version of a) Eulerian model that mainly differ in the mathematical treatment of air parcels and associated chemical interactions

(U.S. EPA 2009). Moreover, modelling resolutions as well as emission, meteorological and population data varied. To ensure comparability, only regional, i.e. European-scale, air quality models are considered. For availability reasons, the Year2013 assessments do not include the updated EMEP (European Monitoring and Evaluation Programme) source-receptor matrices (Schulz et al. 2013), having been used to assess the EU Clean Air Policy Package in 2013.

Table 4.1: Air quality and exposure modelling characteristics for classical air pollutants except ozone for the considered IPA implementations (European Commission 1999b, 2004b, Preiss and Klotz 2008a)

	ExternE1998	**NewExt2004**	**NEEDS2009, Year2013/2013***
Air quality model	Windrose Trajectory (Lagrangian) Model	Windrose Trajectory (Lagrangian) Model	EMEP/MSC-West Eulerian dispersion model, parameterised
Emission inventory	1990	1998	2010 (projected in 2006)
Meteorology	1990	1998	Average of 1996, 1997, 1998 and 2000
Modelling resolution	Eurogrid (100km x 100km)	EMEP50 (50km x 50km)	EMEP50 (50km x 50km)
Population data	EUROSTAT REGIO	EUROSTAT REGIO	SEDAC 2007 and NEEDS

Beyond the changes displayed in Table 4.1, ozone damages were modelled by a generic factor in 1998 and by a site-dependent approach since 2004 (European Commission 2004). The ozone exposure metric was changed from ExternE1998 and NewExt2004, relying on 6-hour average values, to NEEDS2009, being based on the SOMO35 (Sum Of (maximum daily 8h) Means Over 35 parts per billion, ppb) metric, disregarding ozone effects below 35 ppb (European Commission 2005a).

4.3.3 Particle toxicity

Assumptions about the toxicity of different PM compounds relative to primary particles also varied over time: In ExternE1998, sulphates were estimated to be 1.67 times more toxic than nitrates and primary PM_{10} particles because of their typically smaller size and hence larger damage potential (European Commission 1999a, 2005a). In NewExt2004, due to lacking evidence for its effects, nitrates were assumed to be half as toxic as sulphates and primary particles (European Commission 2004). In NEEDS2009 and Year2013/Year2013*, all types of particles are assumed to be equally toxic (WHO 2007, 2013b), although the topic remains debated (cf. section 8.4).

4.3.4 Impact functions

The four IPA implementations use health-related impact functions that are pollutant (i.e. PM_{10}, $PM_{2.5}$ and ozone), risk group and age group-specific (Table 4.2). Driven by new or re-evaluated scientific evidence, changes concerned the impact functions, the affected population fraction or the particle size to which impacts are associated. For comparability reasons, effects that could not be quantified with EcoSenseWeb were disregarded, i.e. SO_2-related endpoints in ExternE1998 and NewExt2004, and NO_2-related endpoints in Year2013/Year2013*. While the latter risks underestimating damage costs, potential double counting with PM impacts and causality issues are debated (WHO 2013b). Since NEEDS2009 and even more so in the Year2013 implementation, endpoints are more frequently related to $PM_{2.5}$ than to PM_{10}. Some endpoints were dropped for Year2013, e.g. bronchodilator usage, and others have been newly introduced, e.g. asthma symptom days among children due to PM_{10}.

Drawing on recommendations by the WHO (2013a), only impact functions with a sufficiently high level of confidence were considered in the 2013 implementations, i.e. categories A* and B* according to the HRAPIE (Health Risks of Air Pollution In Europe) project. For mortality assessment and following HRAPIE, only natural causes are considered, i.e. no deaths due to accidents or other external causes. Two assessment approaches co-exist. While the YOLL (Years Of Life Lost) approach, based on life table calculations (Miller et al. 2011), was generally advocated in the ExternE project series (European Commission 2005a), the U.S. EPA estimates cases of deaths (U.S. EPA 2011a). To account for differing expert opinions, both approaches were used in parallel in the CAFE (Clean Air For Europe) Programme (Hurley et al. 2005) and the impact assessment underlying the EU Clean Air Policy Package (Holland 2014a). For the IPA implementations studied, mortality risks in adults are expressed in YOLL, while for infants cases of death are estimated. The risk coefficient for cases of premature death due to long-term $PM_{2.5}$ exposure as provided by the HRAPIE project needs to be converted into EU average YOLL, consistent with earlier projects. For Year2013 and Year2013*, following an expert recommendation (Hurley et al. 2005), the YOLL-based impact function from NEEDS2009 is scaled linearly using the quotient of the relative risk provided by the HRAPIE (0.062/10 (µg $PM_{2.5}/m^3$) and the NEEDS project (0.06/10 (µg $PM_{2.5}/m^3$)).

Table 4.2: Health-related impact function slopes (s_{CR}) and corresponding risk/age group, expressed as 'effect increase per (10 µg/m³ x person x year)' and grouped by pollutant (European Commission 2004, Torfs et al. 2007, WHO 2013a)

Health endpoint *Risk group, age group*	**Unit**	**ExternE 1998**	**NewExt 2004**	**NEEDS 2009**	**Year2013 /2013***
Ozone[a]					
All-cause, natural mortality (acute)[b]	YOLL				
All, all ages		0.00004	0.00004	0.00002	0.00003
Asthma attack	days				
Asthmatics, all ages		0.4930	0.4930		
Bronchodilator usage (summertime)	cases				
Asthmatics, 20+				0.7300	
Cardiovascular hospital admission (excl. stroke)	cases				
All, 65+					0.0005
Cough day	days				
All, children 5-14				0.9300	
Lower respiratory symptoms (excl. cough)	days				
All, children 5-14				0.1600	
Minor restricted activity day (MRAD)[c]	days				
All, 18-64				0.1154	
All, all ages					0.1201
All, adults 18+		0.0976	0.0976		
Respiratory hospital admission	cases				
All, 65+				0.0001	0.0001
All, all ages		0.00004	0.00004		
Symptom day	days				
All, all ages		0.3307	0.3307		
PM_{10} (primary and secondary)					
All-cause infant mortality	cases				
All, infants 0-1				0.0001	0.0001

Health endpoint *Risk group, age group*	**Unit**	**ExternE 1998**	**NewExt 2004**	**NEEDS 2009**	**Year2013 /2013***
All-cause, natural mortality	YOLL				
All, 30+		0.0072	0.0039		
All-cause, natural mortality (acute)[b]	YOLL				
All, all ages		0.00003	0.00003		
Asthma symptom day	days				
Asthmatic children, children 5-19					1.7374
Bronchitis prevalence					
All, children 6-12	cases				0.0149
Bronchodilator usage	cases				
Asthmatics, 20+				0.9125	
Asthmatics, adults 18+		1.6290	1.6290		
Asthmatics, children 0-18		0.7790	0.7790		
PEACE criteria, children 5-14				0.1825	
Cardiac hospital admission	cases				
All, all ages				0.00004	
Cerebrovascular hospital admission	cases				
All, all ages		0.0001	0.0001		
Chronic bronchitis case	cases				
All, 18+					0.0005
All, 27+				0.0003	
All, adults 18+		0.0005	0.0005		
Chronic bronchitis episode	cases				
All, children 0-18		0.0161			
Chronic cough	cases				
All, children 0-18		0.0207	0.0207		
Congestive heart failure	cases				
All, 65+		0.0002	0.0002		

Health endpoint *Risk group, age group*	Unit	ExternE 1998	NewExt 2004	NEEDS 2009	Year2013 /2013*
Cough day	days				
Asthmatics, adults 18+		1.6760	3.3520		
Asthmatics, children 0-18		1.3350	2.6700		
Lower respiratory symptoms (wheeze)	days				
Asthmatics, adults 18+		0.6060	0.6060		
Asthmatics, children 0-18		1.0290	1.0290		
Lower respiratory symptoms (incl. cough)	days				
All, children 5-14				1.8600	
Respiratory symptoms, adults 15+				1.3000	
Respiratory hospital admission	cases				
All, all ages		0.00002	0.00002	0.00007	
Restricted activity day (RAD); (RAD − net)[d]	days				
All, adults 18+		0.2499; (0.2472)	0.2499; (0.2472)		
$PM_{2.5}$ (primary and secondary)					
All-cause, natural mortality	YOLL				
All, 30+					0.0107
All, all ages				0.0065	
Cardiovascular hospital admission	cases				
All, all ages					0.0002
Minor restricted activity day (MRAD)	days				
All, 18-64				0.5772	
Respiratory hospital admission	cases				
All, all ages					0.0003
Restricted activity day (RAD); (RAD − net)[d]	days				
All, all ages				0.6061; (0.0957)	0.8930; (0.5853)

Health endpoint *Risk group, age group*	**Unit**	**ExternE 1998**	**NewExt 2004**	**NEEDS 2009**	**Year2013 /2013***
Work loss day (WLD)	days				
All, 15-64				0.2070	
All, 20-65					0.6385

[a] Seasonal 6 hour-averages for 1998 and 2004; SOMO35 for 2009 and 2013

[b] Assuming 0.75 YOLL per case for ExternE1998, NewExt2004 and NEEDS2009 (Preiss and Klotz 2008) and 1 YOLL per case for Year2013/2013*(Holland 2014b)

[c] MRAD should be calculated net of asthma attacks due to ozone (European Commission 2004)

[d] To avoid double counting, net effects are obtained by correcting for work loss days, hospital admission days, minor restricted activity days and symptom days due to ozone or PM (Preiss and Klotz 2008, WHO 2013a)

4.3.5 Monetary valuation

Monetary values of health endpoints varied over the years (Table 4.3). Depending on the mortality risk metric, different monetary values are used: YOLL are valued by a so-called Value Of a Life Year (VOLY; previously also abbreviated by VLYL, Value of a Life Year Lost), while cases of death are valued by a Value of a Statistical Life (VSL,) cf. OECD (2012). For the Year2013 implementation, the monetary parameters from the CBA of the EU Clean Air Policy Package were adopted (Holland 2014a). This implies that newer evidence for the adult mortality-related VOLY from the NEEDS2009 project is disregarded. Instead, the parameters recommended in the CAFE Programme in 2005, based on NewExt2004, are used (Holland et al. 2005a). To account for differing expert judgments on this crucial parameter, alternative values for the Year2013 (Holland 2014a) and Year2013* (Desaigues et al. 2011) implementation were used, the latter equalling the NEEDS2009 valuation. Further changes of the Year2013/2013* implementations towards the NEEDS2009 implementation concern:

- All-cause infant mortality: the median VSL based on CAFE implies a decrease of 52% relative to NEEDS;
- Chronic bronchitis cases: an updated value from the Health and Environment Integrated Methodology and Toolbox for Scenario Assessment (HEIMTSA) project (Hunt et al. 2013) is used, amounting only to about 25% of the NEEDS valuation;
- Restricted activity days: the CAFE valuation is used, being 36% smaller than in NEEDS;

- Work loss days: these are valued according to data from the Confederation of British Industry, reducing the NEEDS value by 60%.

For ExternE1998, monetary parameters were converted from ECU_{1995} into $€_{2000}$ using a factor of 1:1 and an average annual inflation rate of 1.5% for the relevant period. For the Year2013/2013* implementations, values were converted from $€_{2005}$ to $€_{2000}$ using an average annual inflation rate of 2.1% (eurostat 2013).

Table 4.3: Monetary values (m) expressed in $€_{2000}$ for health endpoints (European Commission 1999a, 2004, Holland 2014a, Preiss and Klotz 2008)

Health endpoint	ExternE 1998	NewExt 2004	NEEDS 2009	Year2013/ 2013*
all-cause infant mortality (VSL)	-	-	3 000 000	1 442 086[b]
all-cause natural mortality (VOLY, acute exposure)	166 979	75 000	60 000[a]	52 005[b]/ 60 000[a]
all-cause natural mortality (VOLY, long-term exposure)	90 847	50 000	40 000[a]	52 005[b]/ 40 000[a]
asthma attack	81	75	-	-
asthma symptom day	-	-	-	38
bronchitis prevalence	242	-	-	530
bronchodilator usage	40	40	1	-
cardiac hospital admission	-	-	2 000	-
cardiovascular hospital admission	-	-	-	2001
cerebrovascular hospital admission	8 478	16 730	-	-
chronic bronchitis case	113 115	169 330	200 000	48 310
chronic cough	242	240	-	-
congestive heart failure	8 478	3 260	-	-
cough day	8	45	-	-
lower respiratory symptoms (wheeze)	9	8	38	-
lower respiratory symptoms	-	-	38	-
minor restricted activity day (MRAD)	48	45	38	38

Health endpoint	ExternE 1998	NewExt 2004	NEEDS 2009	Year2013/ 2013*
respiratory hospital admission	8 478	4 320	2 000	2 001
(net) restricted activity day (RAD)	81	110	130	83
symptom day	48	45	-	-
work loss day	-	-	295	117

[a] Expressed in €$_{2000}$ (Preiss and Klotz 2008); by contrast, the original study (Desaigues et al. 2011, Desaigues et al. 2007) presumably uses €$_{2005}$, which would reduce the indicated values

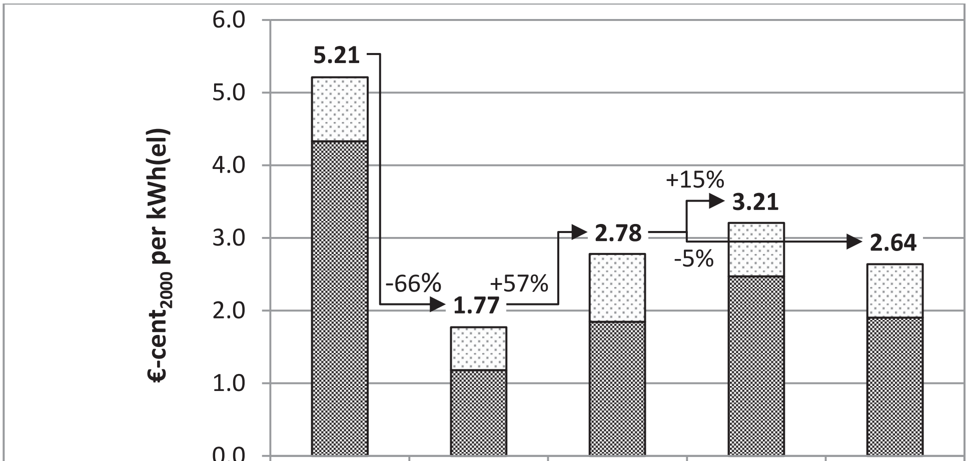
[b] median parameter estimate

4.4 Results

4.4.1 Development of health damage costs over time

The coal-fired power plant's health-related damage costs vary between 1.77 (NewExt2004) and 5.21 (ExternE1998) €-cent$_{2000}$ per kWh of electricity produced (Figure 4.1). Damage costs decreased by 66% from 1998 to 2004, mainly due to reduced mortality impacts, and increased by 57% from 2004 to 2009. Depending on assumptions about mortality impact valuation, estimated damage costs either increase by 15% (Year2013) or decrease by 5% (Year2013*) from 2009 to 2013.

For data availability reasons, only the self-calculated results can be further decomposed. With 66%/77%/70% (NEEDS2009/Year2013/Year2013*) of all quantified damage costs, mortality due to long-term $PM_{2.5}$ exposure is by far the most important single health end-point, consistent with previous observations (Holland et al. 2005b, Krewitt 2002, Spadaro and Rabl 2008). This is followed by work days lost (10%/7%/10%), net restricted activity days (7%/11%/14%), and chronic bronchitis cases (10%/4%/5%).



	ExternE1998	NewExt2004	NEEDS2009	Year2013	Year2013*
Morbidity endpoints	0.88	0.59	0.93	0.74	0.74
Mortality endpoints	4.33	1.18	1.85	2.47	1.90

Figure 4.1: Marginal damage costs of atmospheric emissions from a coal-fired power plant unit, estimated with different IPA implementations

4.4.2 Quantitative analysis on the influence of single elements in the assessment chain

In the following, the influence of individual elements of the assessment chain is studied (Figure 4.2), referring to section 3.4.1, equation 3.3.

Using NEEDS2009 exposure modelling instead of the original exposure modelling reduces the quantified health-related damage costs by 21% and 9.4% for ExternE1998 and New-Ext2004, respectively. This implies that human exposure to classical air pollutants was estimated to be higher in 1998 than in 2004 and 2009 for the studied case. Additionally assuming equal instead of differential particle toxicity reduces damage costs by another 17% for ExternE1998 and increases them by 43% for NewExt2004, underlining the importance of particle toxicity in equation 3.3. When finally using the NEEDS2009 impact function and monetary valuation for long-term exposure mortality, damage costs are reduced by 29% for ExternE1998 and increased by 15% for NewExt2004, bringing both values considerably close to the NEEDS2009 result.

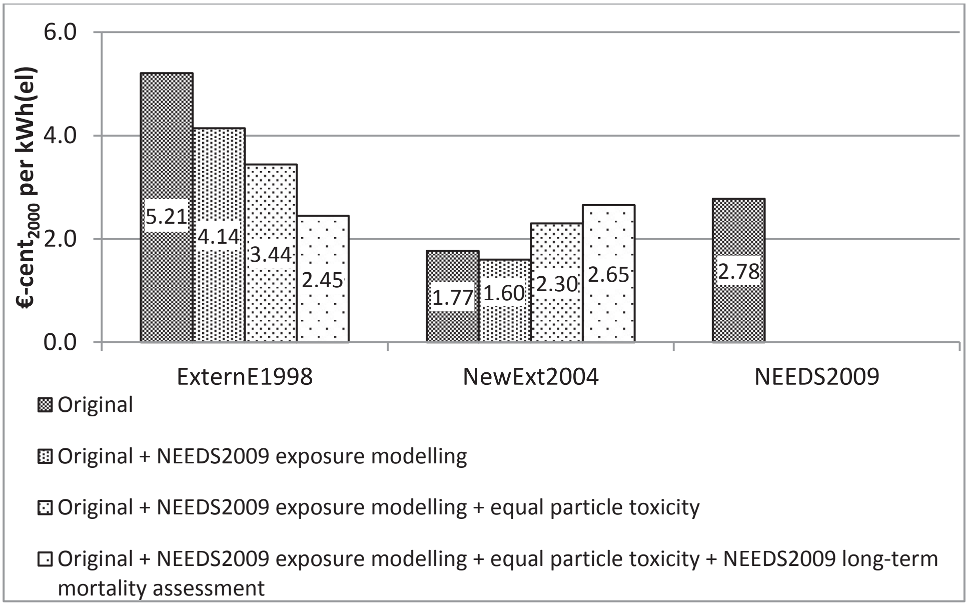


Figure 4.2: Influence of exposure modelling, particle toxicity and long-term mortality assessment on health-related damage costs of atmospheric emissions from a coal-fired power plant unit

This comparison is limited in that the NEEDS2009-based exposure modelling includes neither ozone-related impacts below the 35 ppb concentration threshold nor direct impacts from SO_2. Both types of impacts are included in the original damage costs from ExternE1998 and NewExt2004, explaining part of the residual differences in Figure 4.2.

Given its importance, the development of long-term exposure mortality impact assessment deserves more scrutiny. The original impact function used in 1998 was directly derived from a US study (European Commission 2004). Accounting for differences between Europe and the USA slightly decreased the impact function in 2004. Simultaneously, the associated monetary value was substantially reduced due to a more refined valuation method. From 2004 to 2009, the monetary value decreased further, reflecting updated evidence (Desaigues et al. 2011). This change, however, did not reduce damage costs because of the increased toxicity coefficient for secondary particles. From NEEDS2009 to Year2013, the monetary valuation of long-term exposure mortality impacts increased, while the risk slope remained almost constant. Effectively, an increase of damage costs per concentration increment resulted. In the sensitivity case (from NEEDS2009 to Year2013*), the monetary value remained constant while the risk slope slightly increased, thereby rising mortality-related damage costs.

4.5 Summary of chapter 4

Energy-related health damage cost estimates of four impact pathway approach implementations are compared, corresponding to different stages of the ExternE project series and follow-up activities. Referring to recent European recommendations, two variants of an up to date assessment are proposed, differing only in mortality impact valuation. Disentangling influencing factors that are otherwise hidden in estimates of human health-related damage costs is a central achievement of this chapter, realised through literature research and quantitative analyses. Case study results reveal that exposure modelling accounts for differences between damage costs of up to 21%. Among health endpoints, mortality impacts due to long-term $PM_{2.5}$ exposure are by far the most influential single endpoint (e.g. 76% of the health-related Year2013 damage costs). Particle toxicity assumptions remarkably influence damage costs as well, affecting all particulate matter (PM)-related endpoints simultaneously. PM-related mortality impacts should thus be preferentially studied in sensitivity analyses of health damage costs (cf. section 7.2.2). As a limitation, the quantitative findings cannot be generalised because exposure modelling results and corresponding damage costs depend on the emission pattern and the location of the source. Due to data and model accessibility constraints, the influence of individual exposure modelling components (e.g. spatial modelling resolution) could not be further analysed. However, the analysis of modelling features in section 7.1 provides further insight on this topic.

5 Exemplary application of social cost-benefit analysis using an existing model and weak point analysis

In this chapter, a social cost-benefit analysis (CBA) of emission control at an exemplary coal-fired power plant is carried out. Using and adapting existing assessment models, the feasibility of social CBA at plant level is demonstrated, whilst also allowing to assess the influence of geographic and further settings on damage costs through a sensitivity analysis. The contents are largely based on Bachmann and van der Kamp (2014).

5.1 Introduction

As a result of several European research initiatives in the context of energy externalities, the EcoSenseWeb model enables quantifying environmental impacts in economic terms with a focus on emission reductions related to classical air pollutants. Using the EcoSenseWeb model, a social CBA of an exemplary emission control measure is carried out here, testing the criterion of "disproportionality" in the sense of the EU industrial emissions directive (IED) (cf. section 2.4.6). As was shown for instance by Krewitt et al. (2001), Bachmann (2006), and more recently by Czarnowska and Frangopoulos (2012), the magnitude of external effects varies due to environmental settings. Accordingly, the focus is put on the first criterion for applying for derogations from BAT, i.e. the geographical location or the local environmental conditions. Without loss of generality, the case is made for a typical coal-fired power plant at different Western European locations. Nevertheless, the general approach is applicable also to other kinds of industrial installations as well as to other policy contexts.

5.2 Model: damage cost assessment using EcoSenseWeb

For classical air pollutants, damage costs are estimated on the basis of the impact pathway approach (cf. section 3.4.1), as implemented in the EcoSenseWeb assessment model.

EcoSenseWeb 1.3 (Preiss and Klotz 2008) is used to calculate the environmental benefits resulting from the installation of emission control equipment at an exemplary fossil-fired power plant (cf. section 5.4). The model allows assessing impacts resulting from various emissions into air, comprising classical air pollutants (such as NO_x, SO_2, and primary particles), trace pollutants (such as heavy metals, arsenic, and dioxins) and greenhouse gases. Radionuclide releases and impacts on biodiversity due to land use changes can also be assessed but are disregarded here given the scope of the IED (articles 2 (1), 3 (1a) and 3 (2)). Data and methods necessary to quantify impacts according to the impact pathway approach for releases in Europe are provided. Most of the data is based on results from the EU research project NEEDS (cf. section 4.3 for more details on the models and parameters used). Merely source-specific data need to be entered by the user. Table A.2 (Appendix) gives an overview on all kinds of data that a user can potentially enter. Fewer data may be specified, depending on the scope of the damage costs to be assessed (e.g. regional and/or local; human health, climate change and/or biodiversity). Below, the approaches implemented in EcoSenseWeb for the quantification of impacts from different kinds of air pollutant groups are briefly presented, mainly based on the information provided by Preiss and Klotz (2008).

5.2.1 Classical air pollutants

For classical air pollutants, following the impact pathway approach, impacts on different receptors are considered, i.e. human health, crops, building materials, and biodiversity.

Different dispersion models are available in EcoSenseWeb to assess classical air pollutants in a nested way, covering local, regional (European) and Northern hemispheric impacts. At the local scale, i.e. 100 x 100 km around a point source, a Gaussian plume model calculates dispersion of primary PM only. At the European and Northern hemispheric scale, dispersion as well as chemical conversion is assessed with help of source-receptor matrices derived from the EMEP Eulerian model, involving non-linear relationships. The European modelling domain covers the EU28 plus 11 non EU countries as well as some sea regions. Some larger countries (e.g. France and Germany) are subdivided into sub-regions to allow for a more precise assessment of emissions from these countries.

Different impacts are covered such as mortality and morbidity for human health due to exposure via inhalation, crop yield losses due to airborne pollutants, acidification of soils or fertilising effects, damage to materials due to SO_2 exposure or acid rain, and potential disappearance of plant target species due to acidification and eutrophication.

Monetary values have been obtained through contingent valuation studies (eliciting the Value of a Life Year, VOLY, and monetary values of several respiratory diseases) as well as by relying on market data for crops, treatment costs (e.g. bronchodilator uses), wages (work loss days), and restoration costs (impacts on biodiversity and on building materials). Some impacts occur in the long run. No explicit information on the discount rates used is given in the consulted literature.

In general, receptor data, impact functions and monetary values were determined in different years. Damage costs in EcoSenseWeb are expressed in a common base unit (i.e. $€_{2000}$). Impact functions and monetary values (cf. the NEEDS2009 parameters in Table 4.2 and Table 4.3) stem from different regions, i.e. from Europe and North America.

5.2.2 Trace pollutants

Trace pollutants are assessed to lead to increased human health risks only. Neither dispersion, nor exposure nor impact models for trace elements are implemented directly in EcoSenseWeb. Rather, external unit costs pre-calculated with different models in previous studies are used for trace pollutants. These rely on population data and monetary values that vary from those used in the assessment of classical air pollutants.

5.2.3 Greenhouse gases

Greenhouse gas (GHG) emissions are valued either by a default value, a user-specified constant value, a marginal abatement cost (MAC) or a marginal damage cost (MDC) approach.

When following the MAC approach, monetary values for GHGs are available for emissions occurring between 2000 and 2050. These were derived from a meta-analysis of different models and scenarios by Kuik (2007). The values are valid for a CO_2-eq. concentration target of between 535 and 710 ppm or a CO_2 concentration target of between 440 and 570 ppm. The values in $€_{2005}$/t CO_2 are: 19 until 2020, 23 in 2025 and 61 in 2050 with linear interpolation between these reference years.

EcoSenseWeb allows assessing GHGs up to the year 2100 when using the other approaches, i.e. a default value of 19 $€_{2000}$/t CO_2 (derived based on MAC principles, cf. European Commission (2005a)), a user-specified constant value (of whatever kind), or the

MDC approach. The MDC approach relies on the FUND model[13], developed partly in research projects funded by the European Commission with several publications in the peer-reviewed literature. The MDC values implemented in EcoSenseWeb strongly depend on whether or not the optional equity weighting is used. Equity weighting accounts for the fact that a damage of 1 € is less severe in a highly developed country than in a developing country. Without equity weighting, the values are below 10 $€_{2000}$/t CO_2 while they are two to five times higher than the default value of 19 $€_{2000}$/t CO_2 when considering equity weighting. As a result, the user's choice concerning monetisation of GHG emissions has a pronounced impact on the finally obtained results.

5.2.4 Settings for calculations

Beyond the facility-specific input parameters (Table A.2), EcoSenseWeb offers a few assessment options (cf. Table 5.1). These concern whether or not air quality is assessed at different scales; for regional (i.e. Europe-wide) air quality, the user can choose between two background emission scenarios and between two meteorological conditions; three variants of human health impact functions are provided; impacts on biodiversity are optionally assessed; GHGs and micro pollutants can be included or excluded; for GHG emissions, different monetary values can be specified. For the calculations presented in section 5.4 default settings in EcoSenseWeb were used (Table 5.1). For the sensitivity results, human health impact functions as well as the assumptions regarding air quality modelling have been varied (cf. section 5.4.4).

[13] Cf. http://www.fund-model.org/publications, last accessed: 2017-05-18

Table 5.1: Settings of EcoSenseWeb as used in the analysis of the current chapter; based on Preiss and Klotz (2008)

Setting concerning	Choice	Remarks
Classical air pollutants – dispersion modelling		
Local air quality model	Included	-
Regional air quality model	Included	-
Background emission scenario	2010[d]/2020[s]	Influencing the chemistry of the non-local atmospheric dispersion modelling (non-linearities), anticipated emissions
Meteorological data	Default year[d]/ Future year[s]	Influencing the regional atmospheric dispersion modelling. The default setting is an average of the years 1996, 1997, 1998 and 2000; alternatively, a future setting can be chosen, corresponding to the year 2003
Hemispheric air quality model	Not included	Given the European scope of the IED, hemispheric results are disregarded
Classical air pollutants – impact assessment		
Health risk functions of classical air pollutants	SIA_E_PPM[d]/ SIA_D_PPM_Core[s]	Default risk functions, assuming equal toxicities of different primary and secondary particles; alternatively assuming differing toxicities of primary and secondary particles
Biodiversity losses due to acidification and eutrophication	Included	-
Assessment of other pollutants		
Micro pollutants	Not included	For data availability reasons
GHG valuation	Default value	19 $€_{2000}$/tonne CO_2-eq. (for all options, cf. section 5.2.3)

[d] default assessment; [s] sensitivity analysis

5.3 Analysis of the influence of the geographic location on the damage costs

In the following, a specific investment into an emission control technology is presented that shall be compared to the related environmental benefits, depending on the environmental setting as well as on sensitivities towards specific assumptions. Through a CBA, the overall societal efficiency of the investment at a specific site is tested. While private costs are taken from a publicly available dataset, the EcoSenseWeb model is used to define environmental benefits resulting from the investment.

5.3.1 Case study: retrofitting a coal-fired power plant with a DeNOx system

Installing and operating a selective catalytic reduction (SCR) for reducing NO_x emissions (DeNOx) at the investigated coal-fired power plant changes its technical characteristics (Table 5.2).

Table 5.2: Data used as input into EcoSenseWeb to define the damage costs of a coal-fired power plant unit without and with a selective catalytic reduction (SCR) and primary abatement NO_x measures installed; adapted from European Commission (2004)

	Without SCR	**With SCR**
General data		
Plant type	Existing plant, pulverised hard coal, steam turbine	Existing plant, pulverised hard coal, steam turbine
Thermal input [MW]	1714	1714
Net Electricity sent out [MW]	600	597
Full load hours per year [h/a]	4500	4523
Annual Net electricity generation [GWh/a]	2700	2700
Abatement techniques		
Flue Gas Desulphurisation	installed	Installed
Electrostatic Precipitator	installed	Installed

	Without SCR	**With SCR**
Flue-gas denitrification	n/a	Installed (primary + secondary)
Data relevant for atmospheric modelling		
Stack height [m]	220	220
Stack diameter [m]	5.1	5.1
Flue gas volume stream (full load) [Nm^3/h]	1960000	1960000
Flue gas temperature [K]	363	363
Emissions - classical air pollutants		
SO_2 [mg/Nm³]	252	252
NO_x [mg/Nm³]	820	200
Particulates (PM_{10}) [mg/Nm³]	2.0	2.0
Particulates ($PM_{2.5}$) [mg/Nm³] [a]	1.7	1.7
NH_3 [mg/Nm³]	0.0	0.0
NMVOC [mg/Nm³]	4.0	4.0
Emissions – GHG		
CO_2 [tons/a]	2187000	2197935
CH_4 [tons/a]	27.0	27.1
N_2O [tons/a]	59.4	59.7

[a] As the share of $PM_{2.5}$ in PM_{10} is not known, a value of 86 % is used by default for hard coal-fired power plants operated at full load (Dreiseidler et al. 2000)

The overall efficiency of the plant is reduced by 0.5% due to the SCR's electricity consumption (European Commission 2006b). To produce the same net amount of electricity, the power plant with SCR is assumed to operate some additional hours per year, increasing overall air emissions. The SCR is characterised as follows:

- The SCR, in combination with primary measures, allows to respect the minimum regulatory limit set by the IED (Annex V) for existing coal-fired combustion plants (>300 $MW_{thermal}$) of 200 mg NOx per Nm³ flue gas emitted;

- The flue gas first passes the SCR followed by an electrostatic precipitator for PM reduction and then by a wet flue gas desulphurisation device (high dust configuration), not causing additional NH3 emissions since ammonia is assumed to be absorbed in the electrostatic precipitator used for PM abatement (VGB and EURELECTRIC 2010);

- No additional oxidation module exists at the SCR for reducing non-methane volatile organic compound (NMVOC) emissions;

- Catalyst degradation over time is not considered.

When calculating damage costs, the same power plant unit is assumed to be situated either in Brussels (Belgium), Cartagena (Spain) or Helsinki (Finland, cf. Table 5.3), varying in terms of population density, background emissions and background meteorology. As a result, the parameter values for the two scenarios of Table 5.2 are inserted into EcoSenseWeb three times, i.e. for each of the sites displayed in Table 5.3.

Table 5.3: Geographic coordinates of the three investigated power plant locations

Location	**Brussels**	**Cartagena**	**Helsinki**
Latitude (°N)	51.12	37.59	60.13
Longitude (°E)	4.30	-0.10	24.69

5.3.2 Scope and discounting

Given that CBA theory has been presented in previous chapters of this work (in particular section 3.2), only the most relevant assumptions and data used are presented here.

The decision criterion for the CBA is the net present value. As regards the issue of “who counts” and spatial scope, the individuals considered in the current analysis are those living in the EU as well as some neighbouring countries (i.e. countries for which results are available in the EcoSenseWeb regional assessment). A scope larger than the EU is considered appropriate for two reasons. First, the IED is relevant for the European Economic Area. Second, impacts outside the EU are also of concern to EU policy makers (European Commission 2009). The time horizon for the investment is 20 years according to an assumed remaining power plant lifetime during which the emissions and technical parameters are assumed to be constant.

For determining private costs, the two-stage discounting procedure by Kolb and Scheraga (1990) is used: capital investments are annualised over a given time horizon using an assumed (real) interest rate and then discounted at a (real) social discount rate. Given the Europe-wide scope of the IED, a time-constant social discount rate of 4% is used, recommended in the European Impact Assessment Guidelines (European Commission 2009). While analysing the sensitivity of the chosen discount scheme is generally advisable (Pearce et al. 2006), its choice will not affect the overall result of the current study because the CBA does not compare alternative projects with different cost structures over time.

5.3.3 Determining private costs

Private costs are assessed with an MS Excel spread sheet provided by EGTEI (Expert Group on Techno-Economic Issues (2012))[14], adapted as regards interest rate, equipment lifetime, emission factor without SCR and primary measures as indicated in Table 5.2 and catalyst cost (15 000 € per m^3, cf. (European Commission 2006b)). Following the two-stage discounting procedure described above, capital investments related to the SCR retrofit are annualised over 20 years using an assumed (real) interest rate of 7% and then discounted at the (real) social rate of 4% mentioned above. Results are expressed in $€_{2000}$ per year.

5.3.4 Determining damage costs

The quantified environmental benefits are due to avoided damage costs of air pollution. These are calculated by means of EcoSenseWeb version 1.3 (cf. section 5.2), according to the methodological assumptions specified in section 5.2.4 and the input data presented in Table 5.2 and Table 5.3. EcoSenseWeb results are expressed in €-$cent_{2000}$ per kWh. These are then multiplied by the annual amount of electricity generated in order to yield absolute damage costs in $€_{2000}$ per year for both scenarios at each site.

[14] An international expert group on the costs and technical parameters of power plant technologies; nowadays called TFTEI (Task Force on Techno Economic Issues), see http://tftei.citepa.org/work-in-progress/costs-of-reduction-techniques-for-lcp for further information; last accessed: 2017-05-18. The spreadsheet used here is an earlier version of the cost assessment tool adapted for CBA in chapter 7.

5.4 Results and discussion

In this section, the standard CBA and selected sensitivity analysis results are presented, followed by a discussion on limitations (weak point analysis of the EcoSenseWeb model).

5.4.1 Costs

The annualised capital costs as well as operating costs as shown in Table 5.4 enter the CBA as private costs.

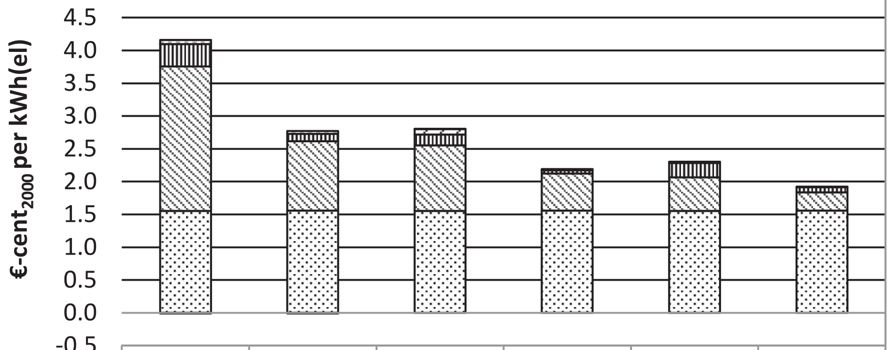
Table 5.4: Private costs of installing a SCR and primary abatement measures at a coal-fired power station, according to EGTEI (2012)

Hard Coal, (grade 2)[a]: existing plant, retrofitted with primary measures and SCR	
Investment costs without catalyst [$€_{2000}$]	25 056 842
Costs for catalyst [$€_{2000}$]	11 842 105
Annualised capital cost [$€_{2000}$ per year]	**3 483 000**
Annual operating costs [$€_{2000}$ per year]	**3 361 012**

[a] Characterised by lower heating value of 24.9 GJ/t and sulphur content of 0.8%, according to EGTEI (2006)

5.4.2 Benefits (avoided marginal damage costs)

According to the default EcoSenseWeb results, the marginal damage costs per kWh produced are reduced through the installation of a DeNOx by 33%, 22%, and 17% at Brussels, Cartagena and Helsinki, respectively (Figure 5.1).



	Brussels_wo	Brussels_w	Cartagena_wo	Cartagena_w	Helsinki_wo	Helsinki_w
Building materials	0.06	0.05	0.01	0.00	0.01	0.01
Crops	-0.01	-0.01	0.08	0.02	0.01	0.00
Ecosystems	0.34	0.11	0.16	0.05	0.22	0.08
Human Health	2.21	1.05	1.00	0.56	0.51	0.28
Climate Change	1.55	1.56	1.55	1.56	1.55	1.56

Figure 5.1: Damage costs of a coal-fired power plant located in Brussels (Belgium), Cartagena (Spain), or Helsinki (Finland) without (wo) and with (w) DeNOx, calculated by means of EcoSenseWeb v1.3 (default assessment)

These results translate into recurring annual benefits as the difference between both scenarios, entering the CBA, as shown in Table 5.5.

Table 5.5: Annual damage costs without (wo) and with (w) DeNOx and resulting annual benefits (default assessment)

	Damage costs wo DeNOx [$€_{2000}$/a]	**Damage costs w DeNOx [$€_{2000}$/a]**	**Annual benefit (wo – w) [$€_{2000}$/a]**
Brussels	112 036 500	74 626 700	**37 409 800**
Cartagena	75 793 900	59 232 500	**16 561 400**
Helsinki	62 218 800	51 930 500	**10 288 300**

5.4.3 CBA results

Entering the annual costs (Table 5.4) and the annual benefits (Table 5.5) into the CBA decision rule (cf. section 3.2) shows that for the standard case, the SCR retrofit can be said to be proportionate for all locations investigated (Table 5.6). The magnitude of the net present value clearly reflects the differences in population density in the affected geographic areas. Reducing emissions at a site that is surrounded by densely populated regions (Brussels) is clearly more beneficial than the same reduction in a region characterised by a low population density. Differences in meteorology also play a role, though the impact on the results is more complex and could not be assessed here.

Table 5.6: Discounted benefits, discounted costs and net present values of the CBA of a SCR installation at a coal-fired power plant at three different locations (default assessment)

	Discounted benefits [$€_{2000}$]	**Discounted costs [$€_{2000}$]**	**Net present value [$€_{2000}$]**
Brussels	528 691 310	96 732 845	**431 958 466**
Cartagena	234 007 154	96 732 845	**137 274 310**
Helsinki	145 395 835	96 732 845	**48 662 990**

5.4.4 CBA results – sensitivity analysis

For the sites with the highest (Brussels) and lowest results (Helsinki), the influence of one technical (full load hours per year; section 5.4.4.1) and three methodological assumptions (particle toxicity, meteorological conditions, and background emissions, section 5.4.4.2) are investigated. Even though EcoSenseWeb allows further settings to be changed, excluding certain models (e.g. on local air quality) or including others (e.g. on hemispheric air quality or micro pollutants) is not deemed appropriate. The sensitivity of GHG emission valuation will not be analysed either despite being a contentious issue for methodological and uncertainty reasons. This study being concerned with NO_x emissions abatement, GHG emissions are hardly affected (cf. Figure 5.1). The impact of changing GHG valuation is thus expected to be limited.

5.4.4.1 Variation in full load hours (operation time)

Changes in European energy markets have led to less base- and mid-load generation of coal-fired power plants (cf. section 2.3). When reducing the annual operating hours from 4500 to 2500, lower benefits from less emission reduction and lower (operating) costs of the SCR result at both sites (Table 5.7).

The CBA results suggest different conclusions for the two investigated sites. At reduced full load hours, a negative net present value is obtained for Helsinki. This result suggests that a short operation time combined with a less densely populated impacted region may make the SCR investment disproportionate. Still, the decrease to 2500 full load hours does not alter the CBA result for Brussels.

Table 5.7: Discounted benefits, discounted costs and net present values for 2500 varying full load hours per year (otherwise default settings)

	Discounted benefits [€$_{2000}$]	Discounted costs [€$_{2000}$]	Net present value [€$_{2000}$]
Brussels	293 748 803	82 359 007	**211 389 795**
Helsinki	80 785 671	82 359 007	**-1 573 335**

5.4.4.2 Varying assumptions on particle toxicity and regional air quality

With a considerable influence on quantified human health costs, the toxicity attributed to different types of particles is scientifically controversial. In the ExternE project series, different particle toxicities have been used (European Commission 1999a, 2004). At EU level and up to now, the WHO recommends considering particles of different sizes, of different composition and from different sources as equally toxic (WHO 2007). Recent evidence suggests that secondary particles originating from the power sector are less toxic than, for instance, primary particle emissions from road traffic (Grahame and Schlesinger 2007). Related to the revision of EU air quality policies in 2013, the WHO (2013b) revisited the question of particle toxicity, pointing out the differing evidence, however without changing its recommendation. Yet, to reflect this uncertainty, the CBA results based on equal particle toxicity are confronted with those based on differing particle toxicity. To this end, the setting in EcoSenseWeb is changed to differential particle toxicity, according to which health effects caused by primary particles (PM_{10} and $PM_{2.5}$), by nitrates (secondary particles formed from NO_x) and by sulphates (secondary particles formed from SO_2) are multiplied by a factor of 1.3, 0.5, and 0.7, respectively.

The discounted benefits decrease substantially under the scenario “different particle toxicity” (Table 5.8). While the DeNOx investment can be considered proportionate for both sites, little changes such as a slight reduction in full load hours at the Helsinki plant, for instance, would alter the CBA outcome. Under future air quality model settings, however, the environmental benefits increase, making disproportionate costs less likely in the scenarios analysed. Note that there are further changes, in particular concerning demography that cannot be considered here.

Table 5.8: Discounted benefits, discounted costs and net present values as a function of default and sensitivity assumptions regarding particle toxicity and air quality modelling

	Particle toxicity	Air quality model setting	Discounted benefits [$€_{2000}$]	Discounted costs [$€_{2000}$]	Net present value [$€_{2000}$]
Brussels	Equal	default	528 691 310	96 732 845	**431 958 466**
		future	*723 811 757*	*96 732 845*	***627 078 912***
	Different	default	262 094 118	96 732 845	**165 361 273**
		future	*434 584 715*	*96 732 845*	***337 851 870***
Helsinki	Equal	default	145 395 835	96 732 845	**48 662 990**
		future	*225 382 624*	*96 732 845*	***128 649 780***
	Different	default	98 685 991	96 732 845	**1 953 146**
		future	*135 817 264*	*96 732 845*	***39 084 419***

5.5 Limitations and weak point analysis

Despite the effort made in selecting representative data and assumptions for the environmental CBA carried out in this chapter, practical and methodological limitations exist, discussed in the following.

5.5.1 Practical considerations regarding the application of the "disproportionate" cost criterion related to industrial emission control technologies

When transposed into Member State legislation the Industrial Emission Directive allows for different implementations of the "disproportionate costs" criterion. Although considering disproportionate costs helps to ensure that societal welfare is enhanced through air quality policies, some countries may decide not to allow for derogations on these grounds. According to the preamble of the IED and without providing more details, "well-defined criteria" shall be taken into account when granting derogations because of disproportionate costs. It will presumably be up to authorities at different levels to define standardised methods. European-wide harmonised standards would allow assessing sector activities across member states in a consistent, non-distortive way. At the time of writing (i.e. end of 2016), consultations with stakeholders, particularly concerned industry sectors, are ongoing at the level of the European Commission, aiming to develop European-wide and possibly sectorial reference methodologies for the assessment of costs and benefits from emission control. Examples of standardisation initiatives already exist by the German Federal Environment Agency (Umweltbundesamt 2012)[15] at national level, or the Clean Air For Europe (CAFE) CBA methodology (Holland et al. 2005a) at European level. However, these recommendations are partly out of date, concerning most notably the monetary value used for assessing the risk of premature mortality (cf. chapter 4). This highlights the need to regularly revise methodological standards according to new scientific findings.

5.5.2 Weak point analysis of the EcoSenseWeb model

Whether the degree of spatial resolution in EcoSenseWeb is sufficient to be applied in the frame of IED derogations regarding "the geographical location or the local environmental conditions of the installation concerned" is a critical and crucial question. The spatial resolution of population data used in the regional model underlying EcoSenseWeb is rather heterogeneous. Likewise, classical air pollutants are not assessed to be emitted at a specific site for the regional scale modelling. Rather these emissions are assumed to occur somewhere in a rather large region (country level or coarse subdivisions of larger countries such as France or Germany). The 50 km grid cell resolution of the regional dispersion model that feeds into EcoSenseWeb does not allow for a proper consideration of effects at the local scale. At the same time, the local dispersion model of EcoSenseWeb, covering an area of 100 km around the emission source, is said to provide "only rough estimates on a coarse resolution", being "more reliable for flat … terrain" (Preiss and Klotz 2008). Moreover, the local model only considers health effects induced by primary PM, thus lacking health impacts due to ozone or secondary particles. Especially in urban areas,

[15] An update of this work with central contributions by this thesis' author regarding the assessment of impacts from classical air pollutants has recently been concluded (van der Kamp et al. forthcoming).

additional approaches are therefore employed in order to better account for higher population densities in exposure modelling when using EcoSenseWeb results, e.g. (Umweltbundesamt 2012, van der Kamp et al. forthcoming).

EcoSenseWeb offers a limited number of assessment options (cf. section 5.2.4). Increasing the degree of freedom through more variable parameter setting options is desirable from a scientific point of view. In particular, in order to integrate scientific advancements, impact functions, and monetary valuation parameters should be directly modifiable[16]. From an application point of view, however, it would be at the expense of user friendliness and also in contrast to the potential strive for a harmonised tool including default settings to be used by authorities across Europe.

EcoSenseWeb does not provide specific information on the estimates' uncertainties. General uncertainty indications related to the health-related damage costs of classical air pollutants are mentioned in the user manual, referring to an assessment of a different air quality model that relies on similar impact functions and monetary values: a typical geometric standard deviation of 3 is suggested for health impacts via inhalation (Spadaro and Rabl 2008), cf. also section 8.2.1. For other impacts assessed above, no uncertainty information is provided. Uncertainty indications being important in decision-making contexts, this lack of detailed uncertainty information in EcoSenseWeb needs remediation.

An important limitation not only of EcoSenseWeb, but also of other external cost assessment frameworks currently available (e.g. the CAFE CBA) is the assessment of biodiversity and ecosystem services-related impacts. In EcoSenseWeb, the potential disappearance of a part of a target plant species for Dutch conditions is used and extrapolated to the rest of Europe (cf. (Ott et al. 2006). At the same time, the current state-of-the-art of biodiversity-related impact assessments concerns the evaluation of changes in ecosystem services (TEEB 2010). In addition, EcoSenseWeb relies on restoration costs for valuation. These can only reliably be used under specific conditions that are rarely met (Bockstael et al. 2000, OECD 2002).

Beyond the limited assessment options and lacking uncertainty consideration, several inconsistencies underlying the methodology used in EcoSenseWeb shall be mentioned. These concern, for instance, different reference years (e.g. meteorological data, receptor

[16] The indirect approach for modification of EcoSenseWeb input parameters employed in chapter 4 is clearly more time and resource consuming than adapting parameters directly within the tool.

data or studies on monetary values), different degrees of geographical validity (e.g. monetary values or risk functions), and different models for assessing concentrations of SO_2 and other classical air pollutants.

Most critically in view of the changing conditions in the European energy sector, the EcoSenseWeb model does not allow distinguishing between different operation schedules. Instead, due to the underlying regional dispersion modelling, constant emissions throughout a given year are assumed. As a result, power plants with relatively low operating hours per year, such as peak-load plants, cannot be dealt with appropriately. Moreover, seasonal peaks of given power plants, such as in the colder winter months, are not well represented. The impacts of peak-load plants with a variable operation pattern throughout the year may thus differ substantially from those assessed by EcoSenseWeb, particularly if effect thresholds are involved (e.g. for ozone). To address this limitation and analyse its importance, a dedicated case study is carried out in section 7.1.3. Besides, the results obtained by using the advanced assessment approach developed in this thesis and those obtained by EcoSenseWeb are compared in section 7.3.2.

5.6 Summary of chapter 5

Using an exemplary case from the power sector, a social cost-benefit analysis is carried out, integrating damage costs from an integrated assessment model, i.e. EcoSenseWeb. The guiding question is whether an emission control measure at a fossil-fired power plant leads to disproportionate costs according to the EU's Industrial Emissions Directive. The chosen example, a DeNOx investment, proves to be socially efficient at all assessed geographic sites. Costs tend to be disproportionate under certain conditions, e.g. lower operating hours per year and remote geographic locations, justifying case-specific derogations. As a result, the techno-economic characteristics of the emission control measures in question and the conditions under which these are installed and operated influence the result of the disproportionate cost evaluation. This shows the need for competent authorities to rely on rather detailed assessments in terms of site and operation conditions when granting derogations due to variable environmental conditions. It is questionable whether the EcoSenseWeb model fulfils this condition when it comes to evaluating impacts from power plants with variable operation profiles, as discussed in the weak point analysis.

6 Development of a cost and benefit assessment methodology for emission control measures at point sources

This chapter, as the methodological core of the thesis, describes how economic assessment approaches are enhanced in order to carry out social cost-benefit analysis (CBA) at the power plant level whilst accounting for variable operation profiles.

6.1 Development of a health damage cost assessment framework

In view of the central objective of the current work, i.e. accounting for the site- and time-dependence of health damage costs, the following requirements are formulated:

- Air quality modelling: considering the precise location of the emission source and the fate of environmental emissions. This includes a highly time- and space-resolved dispersion modelling, accounting for meteorology, chemistry, topography, background emission levels, etc.;

- Exposure assessment: considering population distribution, numbers, and age statistics, preferably in a spatial resolution similar to that of the air quality modelling;

- Health impact assessment: following latest recommendations and using representative input data;

- Monetary valuation: following latest recommendations and using representative input data.

The assessment framework has been developed with these requirements in mind. Following the steps of the impact pathway approach, a marginal damage cost approach is implemented (Figure 6.1).

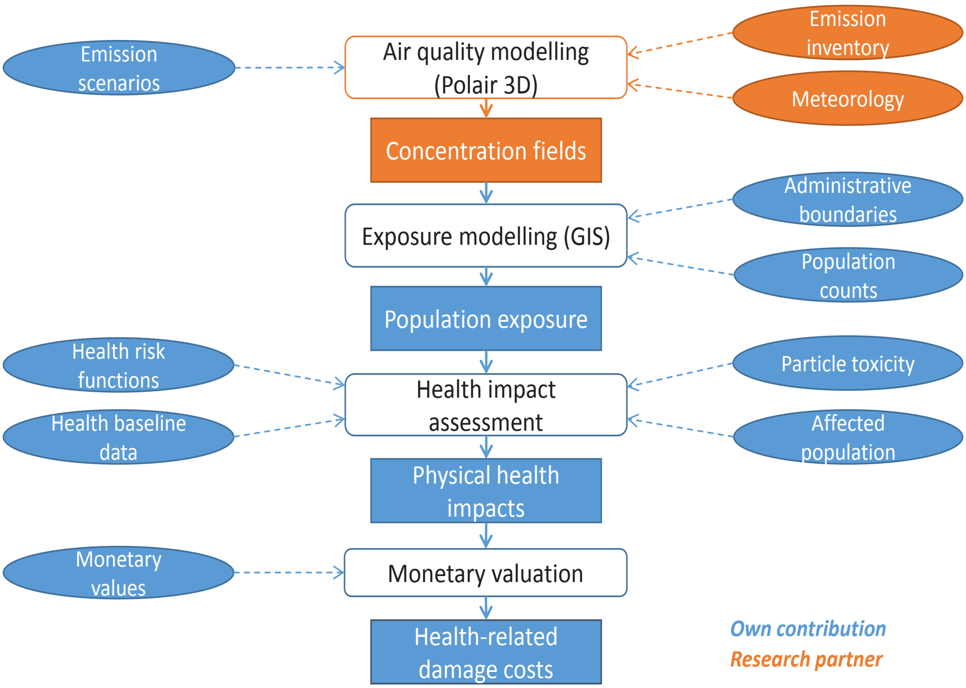


Figure 6.1: Schematic overview of input data, assessment steps and output of the health damage cost assessment framework; own conception based on the impact pathway approach

Atmospheric dispersion modelling has been carried out by CEREA[17] using the Polyphemus platform. The resulting ambient concentration statistics have been integrated into a geographic information system (GIS) for exposure assessment, before proceeding towards a physical and monetary impact assessment. Underlying data processing steps and data sources are summarised in Table 6.1 and specified in the following sections.

[17] Centre d'Enseignement et de Recherche en Environnement Atmosphérique, a French common research laboratory between EDF Research and Development (R&D) and École des Ponts, ParisTech

Table 6.1: Data input and processing steps used in the health damage cost assessment (own compilation)

	Data input	**Data processing steps**
Emission (source) scenario definition	Power plant operation scenarios based on own assumptions; cf. Table 6.2	Scenario construction: defining (hourly) emission rates of air pollutants/pollutant groups (SO_2, NO_x, and PM) during one year
Atmospheric modelling	Power plant emission scenarios, background emission inventory and meteorology; cf. section 6.1.1, Table 6.4	Chemical transport modelling: carried out by CEREA using the Polyphemus platform (Mallet et al. 2007)
Exposure assessment	Hourly concentration fields of PM_{10}, $PM_{2.5}$, and ozone (O_3) (Polyphemus output data) Population counts and administrative boundaries; cf. section 6.1.2, Table 6.5	Data extraction, i.e. extracting concentration fields from binary files and importing them into geodatabases: defining headers (including information on the geographic extensions, the grid cell resolution and number format amongst others) and geographic projections (possibly using geographic transformations) to generate raster data; Zonal statistics: using “resampling” (generating a very fine grid cell resolution, depending on the size of the corresponding administrative units) and “zonal statistics as table” functions in ArcGIS® 10.3 in order to define average concentration levels per administrative unit (France: commune level; Europe: NUTS3 level); storage of generated data in table format; Exposure calculation: merging information on spatially and temporally resolved concentration changes with spatially resolved population counts.
Health impact assessment	Age group data, risk group data, health baseline data, and risk functions; cf. section 6.1.3, Table 6.6 and Table 6.7	Merging data on population exposure with health impact functions; specific approach for assessing long-term exposure mortality risks in adults using life table methods

	Data input	Data processing steps
Monetary valuation	EU-average parameters adapted from Holland (2014a); cf. section 6.1.4, Table 6.8	Merging information from health impact assessment with monetary values

6.1.1 Air quality modelling

6.1.1.1 Definition of emission scenarios

Different operation scenarios and corresponding emission levels at power plant level were defined (Table 6.2) according to the research objectives underlying this work, i.e. advancing the assessment methodology of emission control measures at fossil fuel power plants with variable operation profiles.

These scenarios enable the analysis of methodological aspects (e.g. influence of spatial and temporal modelling resolution) as well as the application of CBA at site-level, accompanied by sensitivity analyses (cf. chapter 7). Note that only the operation phase of the power plant is considered, while related up- and downstream activities are not deemed relevant here. In order to illustrate the annual operation pattern of the power plant units in the scenarios S1, S2, and S3, the daily SO_2 emissions of scenario S2 are displayed (Figure 6.2). Further relevant characteristics of the power plant are as follows: heavy fuel oil-fired power plant, boiler, consisting of 4 units and located in Western France. The stack height is assumed to be 220 m and the stack diameter 5.37 m.

Table 6.2: Power plant operation scenarios and related emission levels, valid for units 1-4 of the power plant respectively; parameter changes from one scenario to the next in bold; own assumptions

Scenario	Full load hours (eq.)	Elec. capacity per unit	Flue gas volume stream	SO_2 concentration	SO_2 emitted	NO_x concentration	NO_x emitted	PM concentration	PM emitted
S0	Units 1-4 of the power plant do not operate (baseline scenario)								
S1	500 h/a	600 MW_{el}	1800000 Nm^3/h	Fuel with 0.33% sulphur content; 600 mg/Nm^3	540 t/a	Initial state: 600 mg/Nm^3	540 t/a	Initial state: 40 mg/Nm^3	36 t/a
S2	500 h/a	600 MW_{el}	1800000 Nm^3/h	**Fuel with 0.22% sulphur content: 400 mg/Nm^3**	**360 t/a**	**Low-NOx burner; 400 mg/Nm^3**	**360 t/a**	Initial state: 40 mg/Nm^3	36 t/a
S2b	**8760 h/a**	**34 MW_{el}**	1800000 Nm^3/h	Fuel with 0.22% sulphur content: 400 mg/Nm^3	360 t/a	Low-NOx burner; 400 mg/Nm^3	360 t/a	Initial state: 40 mg/Nm^3	36 t/a
S2c	8760 h/a	**600 MW_{el}**	1800000 Nm^3/h	Fuel with 0.22% sulphur content: 400 mg/Nm^3	**6307 t/a**	Low-NOx burner; 400 mg/Nm^3	**6307 t/a**	Initial state: 40 mg/Nm^3	**630.7 t/a**
S3	**500 h/a**	600 MW_{el}	1800000 Nm^3/h	Fuel with 0.22% sulphur content: 400 mg/Nm^3	**360 t/a**	Low-NOx burner; 400 mg/Nm^3	**360 t/a**	**Electro static precipitator: 20 mg/Nm^3**	**18 t/a**

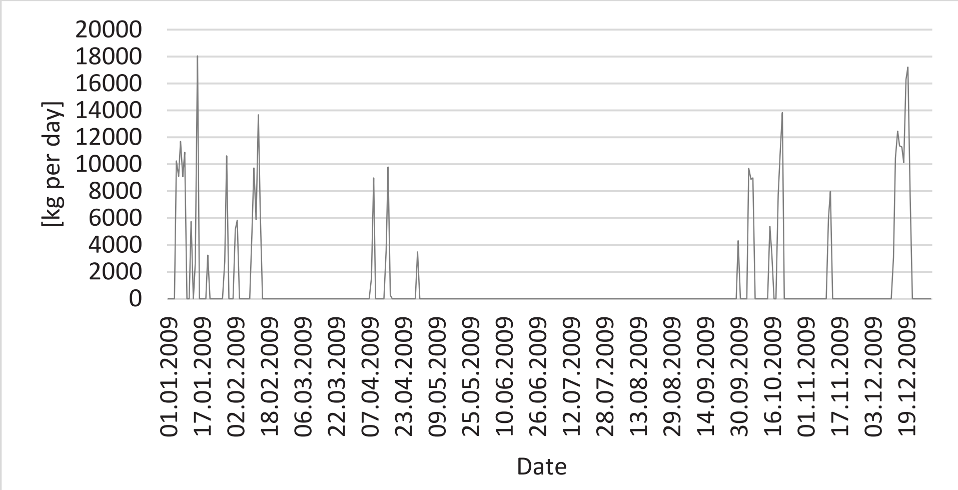


Figure 6.2: Daily SO_2 emissions per unit in scenario S2, reflecting the operation profile of the oil-fired power plant throughout the year; own compilation

6.1.1.2 Atmospheric modelling

Table 6.3 summarises elements of the scope definition related to atmospheric dispersion modelling and illustrates the scope of the methodology developed in this thesis.

Table 6.3: Elements to be considered in the scope definition of atmospheric modelling and specific choices made in the current thesis (grey shaded); own compilation

Emission source	point source	line source	area source	
Considered pollutants	primary	secondary		
Spatial extension	local	national	regional	global
Spatial resolution	high	medium	low	
Time extension	day	month	season	year
Time resolution	hour	day	month	year
Air quality modelling	simple box model	Gaussian	Eulerian	
Modelling approach	averaged	parameterised	specific	

As regards the spatial scope, ambient air pollution concentration data related to the different emission scenarios (Table 6.2) at three scales were obtained:

1) Local domain, i.e. *Île-de-France* region and parts of surrounding regions (bottom-left of Figure 6.3);

2) National domain, i.e. mainland of *France* and Corsica (top-right of Figure 6.3);

3) European domain, i.e. large parts of *Europe*[18] (Figure 6.4).

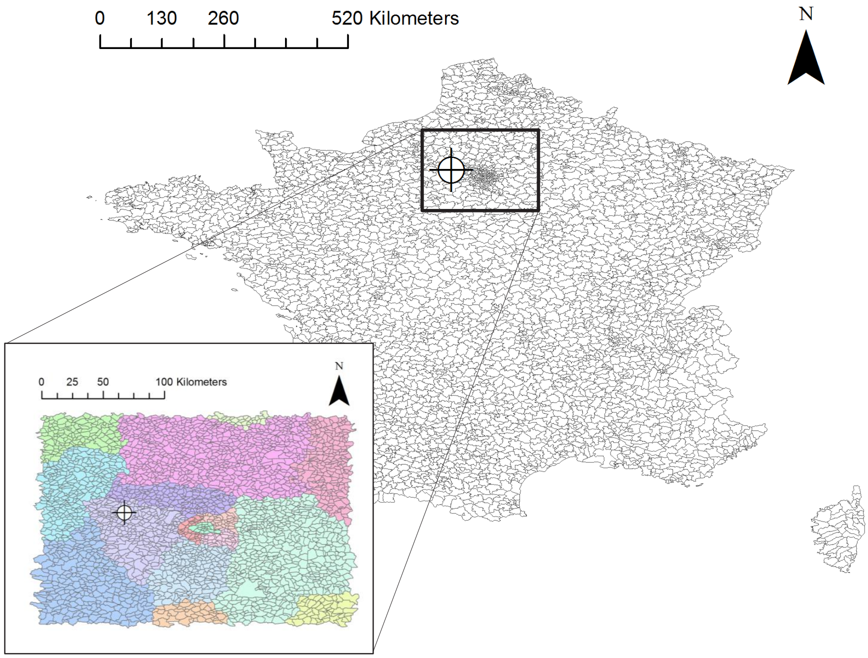


Figure 6.3: Modelling domain "France", displaying the "canton"[19] level, with enlarged section of "Île-de-France" modelling domain, displaying "communes" per (colour-differentiated) NUTS3 zone; emission source located at white dot; maps based on IGN (2013)

[18] Although this domain includes parts of Bosnia and Herzegovina (mapped without background colour fill), these countries are excluded from the analysis due to lacking administrative data. The same applies to Andorra and San Marino.

[19] Displaying the "commune" level at national scale is not helpful here - the figure would become unreadable due to the over 36 000 communes on the territory.

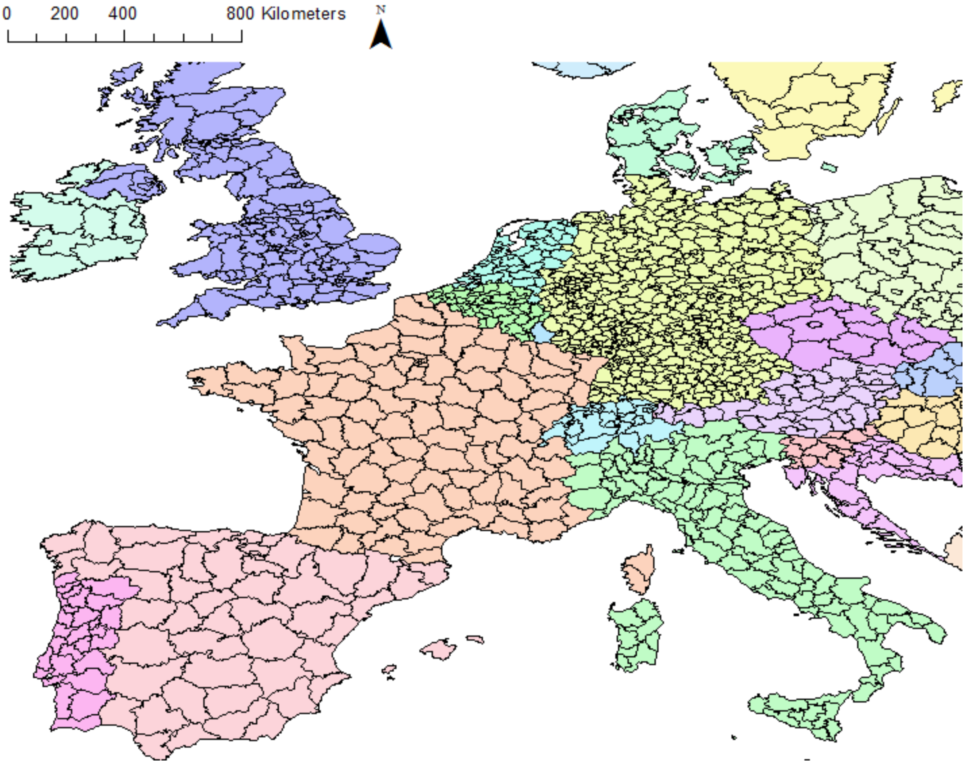


Figure 6.4: Modelling domain "Europe", displaying NUTS3 administrative zones per (colour-differentiated) country; map based on eurostat (2015a), © EuroGeographics for the administrative boundaries

The background emission and meteorological data of the year 2009 were used in the air quality modelling, as its meteorological conditions[20] are considered representative for current day analyses, i.e. marked by increasing average temperatures when compared to the historic long-term average rates.

In preparing the WRF meteorological data, a so-called 1-way nesting was used, meaning that meteorological boundary conditions from a higher-level scale are passed on towards the lower-level in order to obtain more accurate simulation results at the borders of the respective domains.

[20] Cf. http://www.meteofrance.fr/climat-passe-et-futur/bilans-climatiques/autres-annees/bilan-de-lannee-2009, last accessed: 2017-05-18

The atmospheric dispersion modelling was carried out using the software package Polyphemus (Mallet et al. 2007). Detailed specifications and input data are summarised in Table 6.4.

Table 6.4: Characteristics and data sources of the atmospheric modelling; based on Legorgeu (2016)

	Île-de-France	France	Europe
Air quality models used (modules within Polyphemus platform)	Polair3D (Eulerian model), "plume-in-grid" modelling (Gaussian)	Polair3D (Eulerian model), "plume-in-grid" modelling (Gaussian)	Polair3D (Eulerian model), "plume-in-grid" modelling (Gaussian)
Geographic coordinates of domain (decimal degrees)	1.2°E - 3.4°E/ 48.2°N - 49.69°N	5°W - 9.84°E/ 41.2°N - 51.32°N	10.2°W - 19.03°E/ 36°N - 58.49°N
Horizontal and vertical grid cell resolution at surface	0.045° (~3 km x ~5 km)	0.2249° (~17 km x ~25 km)	0.4497° (~35 km x ~50 km)
Input: emission source characteristics	Oil-fired power plant, 4 units; operation profiles accord. to Figure 6.2		
Input: emission inventory - year	European Monitoring and Evaluation Programme (EMEP) - 2009/EMEP - 2020 (projected scenario)		
Input: meteorology - year	WRF 3.6.1 (Weather Research and Forecasting Model 2015) - 2009		
Input: land cover data	Global Land Cover 2000 database (Hartley et al. 2006)		
Output: concentration fields	Substances/substance groups available: NO, NO_2, SO_2, O_3, $PM_{2.5/10}$ (primary/secondary (organic/inorganic) fractions)		
Output: time coverage	365 days, 1-hour time steps		

Polair3D (Boutahar et al. 2004), the main module used, is an Eulerian chemistry-transport model that accounts for advection (transport of air by wind), turbulent diffusion (random and chaotic time-dependent motions of air, mainly vertical) and chemical transformations, involving gaseous and aqueous chemistry (Figure 6.5).

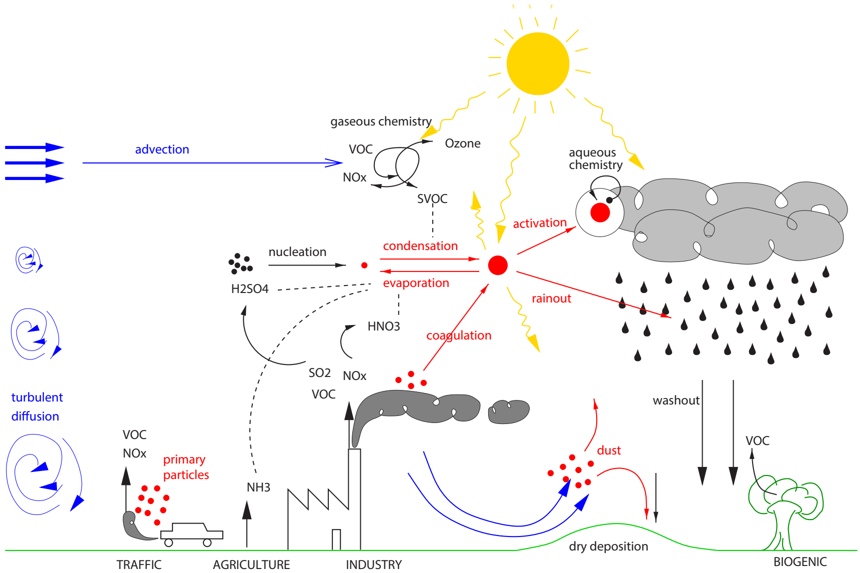


Figure 6.5: Atmospheric processes considered in Eulerian chemistry-transport models; reprinted with permission from and based on Sportisse (2007)

Differential equations within 3-dimensional grid cells are used to estimate the fate of aerosols and other substances in the atmosphere. It builds on the SIREAM (SIze REsolved Aerosol Mode) model for defining size distributions (Debry et al. 2007), distinguishing 17 aerosol species and treating effects such as condensation, evaporation, coagulation, and nucleation. In a post-processing step and using the models ISORROPIA (Nenes et al. 1998) and H^2O (Couvidat et al. 2012) respectively for secondary inorganic aerosols and secondary organic aerosols, Polair3D allows to consider the formation of secondary PM species. For the current application, a modelling interval of 600 seconds is used, but only every 6th time-step, equivalent to one hour, is extracted as output concentration field in order to limit the overall data amount. Eight vertical layers are considered in the modelling used here (Legorgeu 2016), of which only the lowest layer (i.e. ambient air) is relevant for health impact assessment.

In a complementary analysis and for a more robust dispersion modelling around the point emission source, a so-called "plume-in-grid" modelling is applied (cf. section 7.1.4), further described by Kim et al. (2014) and Korsakissok and Mallet (2010). This means that a Gaussian model is embedded within the Eulerian model domain, simulating the dispersion of the emission plume near the source at higher temporal and spatial resolution. As a consequence, the diffusion and dispersion of the pollutants are modelled with a higher precision before eventually injecting the concentration field data into the Eulerian modelling grid. The following specifications for the "plume-in-grid" modelling were used:

- Modelling time frequency: 200 seconds;
- Criterion used for injecting puff data from Gaussian into Eulerian model: either as soon as the horizontal size of the puff reaches the size of the Eulerian grid cell size or, if the first condition is not met, after a duration of 1 hour (puff travel time);
- Use at the metropolitan and national modelling domain, given that the effect at the European scale is expected to be limited.

Apart from verifying the consistency of the modelling outputs, a validation of modelled concentrations with real-world monitoring air quality data was not carried out in this thesis, given that this subject has already been addressed elsewhere, e.g. by Tombette and Sportisse (2007), Zhang et al. (2013), Lecœur and Seigneur (2013) or Kim et al. (2014). According to these, the Polyphemus models predict concentrations of ozone and of overall $PM_{2.5}$ rather well, whereas the results for secondary PM in terms of nitrates and sulphates are less robust (cf. also section 8.4.1).

6.1.2 Exposure assessment

Exposure assessment-relevant data and sources are summarised in Table 6.5.

Table 6.5: Data used for exposure assessment (own compilation)

	Île-de-France	France	Europe
Population counts: level of detail and data source	Commune, year 2009 (INSEE 2009b)	Commune, year 2009 (INSEE 2009b)	NUTS3, year 2009 (eurostat 2015c)
Administrative boundaries: level of detail and data source	Commune, year 2011 (IGN 2013)	Commune, year 2011 (IGN 2013)	NUTS3, year 2010 (eurostat 2015a) © EuroGeographics

Administrative boundaries are classified according to their NUTS (Nomenclature of Territorial Units for Statistics) level, a classification of geographic units established by the European Union[21]. NUTS3 refers to small regions, e.g. the “département” level in France. For surrounding countries, the sizes of the NUTS3 regions differ (cf. Figure 6.4), depending mainly on population numbers. Below NUTS3, local administrative units (LAU) exist, notably LAU2 which corresponds to the “communes”, i.e. commune level in France (Figure 6.3).

A key element during exposure modelling is the spatial distribution of the population and the density per modelling grid cell. Below, the spatial distribution in France per commune is displayed (Figure 6.6).

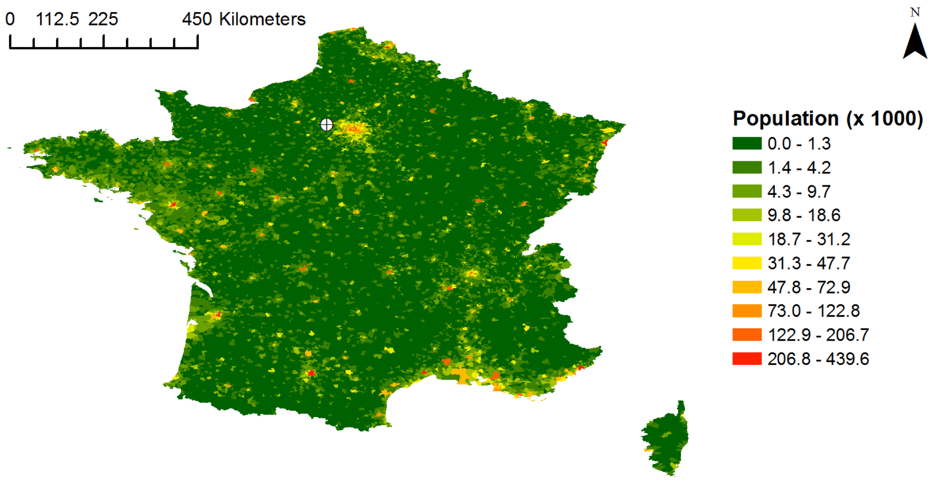


Figure 6.6: Population number per commune in France in 2009 and emission source location (white dot); based on IGN (2013)

In order to combine the concentration raster data coming from the atmospheric modelling with population data that is available per administrative unit, the “zonal statistics” function, as implemented in ArcGIS version 10.3, is used. Its principle is displayed in Figure 6.7. In a first preparatory step, the raster data is resampled in order to obtain a higher resolution raster. The zonal statistics function then calculates the average value

[21] http://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1424956678004&uri=URISERV:g24218, last accessed: 2017-05-18

per administrative unit based on the raster data that is overlapping the respective administrative units. In case of several raster cells overlapping one administrative unit, the weighted average of the raster data is calculated.

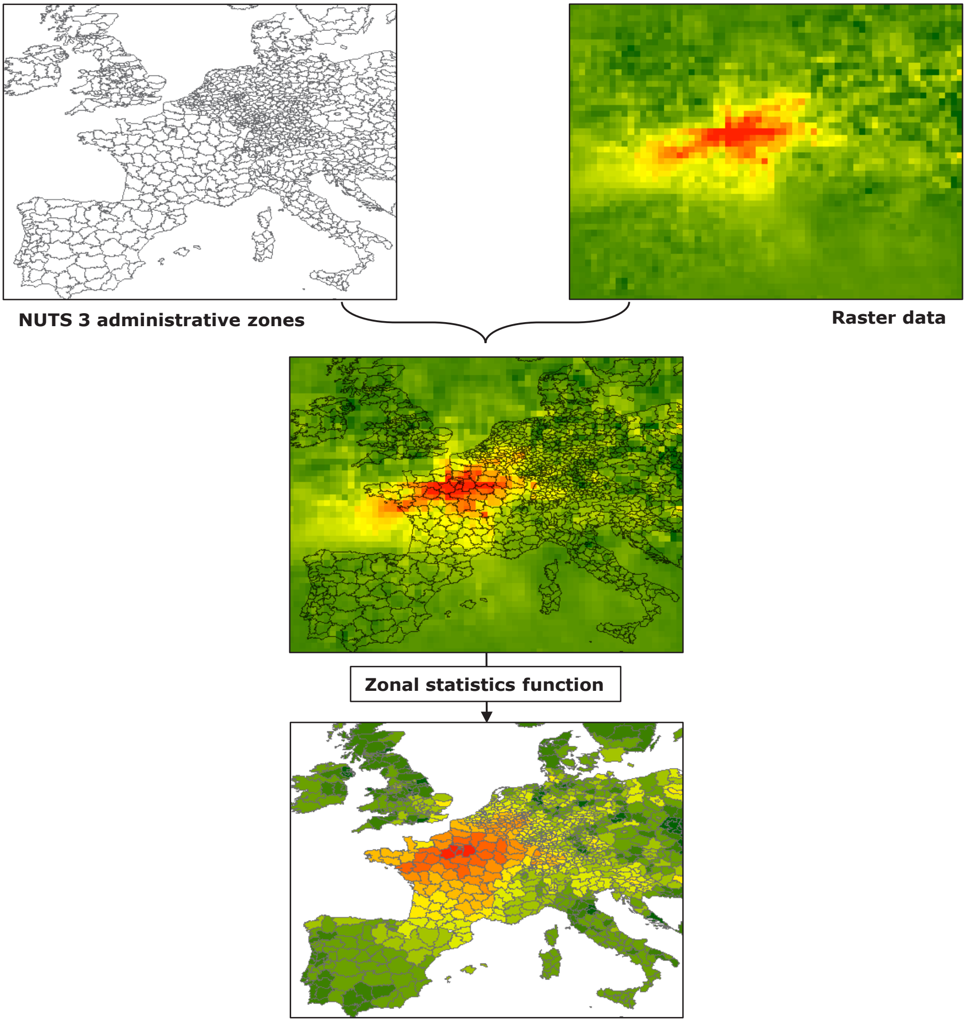


Figure 6.7: Data processing during exposure assessment, featuring the zonal statistics function; exemplary scheme for the European domain (own compilation)

6.1.3 Health impact assessment

Health impact assessment aims at estimating physical health endpoints, using risk parameters, baseline rates and the affected population fractions. Unlike atmospheric modelling and exposure assessment that are carried out in a highly site-specific way, health impact assessment partly relies on EU-average data for both availability and consistency reasons (Table 6.6).

Table 6.6: Data used for health impact assessment at the three assessment domains (own compilation)

	Île-de-France	**France**	**Europe**
Age group data	Ile-de-France data, year 2009 (INSEE 2009c)	French average data, year 2009 (INSEE 2009c)	EU-average data, year 2009 (WHO 2016)
Health baseline data	French average data, year 2009 (WHO 2016)	French average data, year 2009 (WHO 2016)	EU-average data, year 2009 (WHO 2016), except for long-term exposure mortality assessment
Health risk functions	EU average data according to WHO (2013a), cf. Table 6.7		

6.1.3.1 Parameter selection for health impact assessment

Numerous studies have studied the links between exposure to ambient air pollutants and changes in health status of the concerned population. This has led to a large variety of available health risk functions for many different endpoints.

For European assessments, reference datasets have been regularly compiled, e.g. in research projects related to the ExternE project series or by the World Health Organization (WHO), cf. also chapter 4. These respectively reflect the best available scientific evidence of their time. The last concerted effort at European level was the HRAPIE (Health Risks of Air Pollution In Europe) project by the WHO that recommends health impact functions for use in regulatory cost-benefit analysis (WHO 2013a) based on an extensive meta-analysis. Since these recommendations reflect the latest and most robust scientific evidence for European boundary conditions, it was decided to use them as basis for health impact assessment. Table 6.7 summarises health endpoints and associated data used by default in the damage cost assessment. It includes a quality metric (Q), differentiating between two groups of endpoints (WHO 2013a):

- A*: pollutant–outcome pairs for which enough data are available to enable reliable quantification of effects;

- B*: pollutant–outcome pairs for which there is more uncertainty about the precision of the data used for quantification of effects.

The asterisk (*) denotes that these endpoints are additive, i.e. they can be aggregated without risking double-counting of effects. For the default impact assessment, only effects from groups A* and B* are considered, noting that further, less robust, endpoints exist.

A double counting risk remains regarding NO_2 endpoints and corresponding endpoints related to $PM_{2.5}$ or PM_{10}; therefore NO_2-related effects are currently not considered by default (indicated through grey colour in Table 6.7). However, a sensitivity analysis that includes NO_2-related effects is carried out, cf. section 7.2.1.

During data retrieval and compilation, recommendations from Holland (2014b), related to the WHO's HRAPIE project, were largely followed, while noting that:

- Net restricted activity days are defined as follows (Torfs et al. 2007, WHO 2013a): NetRAD = RAD – WLD – (RHA and CHA)*10 days - (asthma symptom days*(5/3));

- Work loss day statistics: EU average baselines were used since no up-to-date data was available for France in the WHO Health For All Database (HFA DB); employment rate taken from eurostat (2015b).

As regards the concerned population fraction, age group data is displayed in Table A.3 (Appendix) and for the only risk group fraction distinguished, i.e. asthmatics among children, the average prevalence provided by the WHO for Western Europe (4.9 %) is used throughout the European modelling domain. Even though WHO recommends a different rate in Northern and Eastern Europe, this has not been considered here for practical reasons and the relatively minor importance of this endpoint. Baseline rates for France and Europe are given in Table A.4 (Appendix).

Table 6.7: Health endpoints and data used in the assessment framework developed in this thesis; NO_2-related endpoints (grey) for sensitivity analysis; based on WHO (2013a)

Q	Pollu-tant	Metric	Health endpoint	Risk group	Age group	CRF [1/(10µg/m³)]	Baseline (BL) per person (EU average)		Source of baseline	s_{CR} (=CRF*BL) [1/(10µg/m³)* person*a]	
A*	$PM_{2.5}$	Annual mean	All-cause natural mortality	All	Adults30+	0.0620	0.0146	Deaths/year	WHO DMDB	0.00980[a]	YOLL
A*	$PM_{2.5}$	Daily mean	Cardiovascular hospital admission	All	All	0.0091	0.0240	Cases/year	WHO HMDB	0.00022	Cases
A*	$PM_{2.5}$	Daily mean	Respiratory hospital admission	All	All	0.0190	0.0139	Cases/year	WHO HMDB	0.00026	Cases
B*	$PM_{2.5}$	Annual mean	Net restricted activity day (netRAD)	All	All	0.0318	19	Days/year	(Ostro and Rothschild 1989)	0.88818	Days
B*	$PM_{2.5}$	Annual mean	Work loss day (WLD)	All	Adults20 to64	0.0460	9.8739	Days/year	WHO HFA DB, (eurostat 2015b)	0.45420	Days
B*	PM_{10}	Daily mean	Asthma symptom day	Asthm. children	Children5 to19	0.0280	0.1700	Daily inci-dence	(WHO 2013a), ISAAC study	1.73740	Days
B*	PM_{10}	Annual mean	All-cause infant mortality	All	Infants	0.0400	0.0014	Deaths/year	WHO HFA DB	0.00006	Deaths
B*	PM_{10}	Annual mean	Bronchitis prevalence	All	Children6 to12	0.0800	0.1860	Annual preva-lence	(WHO 2013a), PATY study	0.01488	Cases
B*	PM_{10}	Annual mean	Chronic bronchitis case	All	Adults18+	0.1170	0.0039	Cases/year	(WHO 2013a), SAPALDIA study	0.00046	Cases
A*	O_3	SOMO35	All-cause natural mortality (acute)	All	All	0.0029	0.0108	Deaths/year	WHO MDB	0.00003[b]	YOLL
A*	O_3	SOMO35	Cardiovascular hospital admission (excl. stroke)	All	Adults65+	0.0089	0.0578	Cases/year	WHO HMDB	0.00051	Cases
A*	O_3	SOMO35	Respiratory hospital admission	All	Adults65+	0.0044	0.0297	Cases/year	WHO HMDB	0.00013	Cases
B*	O_3	SOMO35	Minor restricted activity day (MRAD)	All	All	0.0154	7.8	Days/year	(Ostro and Rothschild 1989)	0.12012	Days
A*	NO_2	Daily 1-h max	All-cause natural mortality (acute)	All	All	0.0027	0.0108	Deaths/year	WHO MDB	0.00003	Deaths
A*	NO_2	Daily mean	Respiratory hospital admission	All	All	0.0180	0.0139	Cases/year	WHO HMDB	0.00025	Cases
B*	NO_2	Annual mean, > 20 µg/m³	All-cause, natural mortality	All	Adults30+	0.0550	0.0146	Deaths/year	WHO DMDB	0.00980[a]	YOLL
B*	NO_2	Annual mean	Bronchitis symptom days	Asthm. children	Children5 to14	0.0210	0.2990	Annual preva-lence	(WHO 2013a)	2.29184	Days

[a] Based on simplified YOLL approach, described above; at the French level, a detailed life table approach is used, equally described above

[b] Assuming 1 YOLL per death (Hurley et al. 2005)

Abbreviations used: Q: quality metric, as explained above; CRF: concentration-response function; DMDB (Detailed Mortality Database); HFA DB (Health For All Database); MDB (Mortality Database); HMDB (Hospital Morbidity Database); SOMO35 (Sum Of Means Over 35 ppb)

6.1.3.2 Assessing long-term exposure mortality impacts

Long-term exposure mortality impacts due to $PM_{2.5}$ (and NO_2, used in a sensitivity analysis) merit special attention, given that this endpoint regularly represents the highest share among quantified health-related damage costs (Holland 2014a). Two assessment approaches can be distinguished (WHO Regional Office for Europe and OECD 2015):

- Quantification is based on years of life lost (YOLL), which are subsequently valued by a so-called Value of a Life Year (VOLY);
- Quantification is based on the number of premature deaths, which are subsequently valued by a so-called Value of a Statistical Life (VSL).

While infant mortality is quantified using the number of premature deaths, mortality in adults due to O_3, $PM_{2.5}$, and NO_2 is expressed as YOLL, requiring additional assumptions and data as explained in the following. A quantification of premature deaths as an alternative to YOLL is presented in section 7.2.2.

Life table method approach to estimate mortality impacts due to long-term $PM_{2.5}$ and NO_2 exposure

The life table (also called survival curve) method aims at estimating the number of life years lost (gained) related to changes in ambient air pollution levels by accounting for changes in population numbers and mortality rates over time. It is based on national life expectancy data and mortality baseline rates per age group, therefore considering the age of death in the assessment. By relying on age group-specific data, it considers the susceptibility, e.g. of elderly people, to die prematurely due to air pollution risks, a factor that cannot be easily considered when quantifying cases of death. At the same time, the fact of implicitly assigning fewer life years lost to the death of elderly people is subject to ethical objections, as this could lead to age discrimination (OECD 2016). Notwithstanding these objections, the EU has chosen the YOLL-based approach as standard when estimating air pollution-related impacts in adults (Hurley et al. 2005), whereas the U.S. EPA by default bases their impact assessments on cases of death without considering the associated lifetime lost (U.S. EPA 2011a).

For the mortality risk assessment of the current thesis, a life table approach based on Miller (2013) was implemented, using life table templates combined with French age group data and mortality statistics (INSEE 2009a, c). For the rest of Europe, an alternative approach is used, explained below.

- Approach and assumptions for life table calculation: The impact of a 1-year pulse reduction of 1 µg/m³ ambient PM2.5 (or NO_2) concentration on the all-cause mortality rate of adults aged 30+ in France is studied, using the concentration-response functions currently recommended by the WHO (cf. Table 6.7) and a discount rate of 0%, consistent with Hurley et al. (2005). It is assumed that effects occur with a given time lag, i.e. 30% of the mortality rate reduction in year 1, 50% during years 2 to 5 and the remaining 20% during years 6 to 20, consistent with a lag scheme by the U.S. EPA (2011a).

- Results: By cumulating the years of life gained for the entire cohort over a period of more than 100 years (starting in year 2009) the life years gained across the French population aged 30+ are obtained. For $PM_{2.5}$, this number amounts to 41 407 life years gained per 1 µg/m³ ambient $PM_{2.5}$ concentration reduction, whereas for NO_2, due to the somewhat lower risk coefficient, it amounts to 36 856 life years gained per 1 µg/m³ ambient NO_2 concentration reduction. Considering the relevant population statistics, this translates into 64.08 (101.79) YOLL per 100 000 people all ages (aged 30+) per 1 µg/m³ ambient $PM_{2.5}$ concentration increase or 57.04 (90.6) YOLL per 100 000 people all ages (aged 30+) per 1 µg/m³ ambient NO_2 concentration increase during 1 year.

These figures, used hereafter as reference for the mortality risk assessment at French national level, are close to the European average of 65.2 YOLL per 100 000 people (all ages) per 1 µg/m³ ambient $PM_{2.5}$ concentration increase during 1 year, recommended by the CAFE and NEEDS projects (Hurley et al. 2005).

For illustrative purposes, an alternative scenario has been calculated, assuming that a 1 µg/m³ reduction in $PM_{2.5}$ concentration is sustained over 20 consecutive years (equivalent to the assumed power plant lifetime in the CBA case study, cf. section 7.3.1). This scenario leads to 1 338 YOLL avoided per 100 000 (all ages) for a 1 µg/m³ ambient $PM_{2.5}$ concentration decrease during 20 consecutive years and exceeds the result of the 1 year pulse change multiplied by 20 (20*64.08 = 1280). This is because the 1 year pulse approach does not account for changes in the population at risk over the years (i.e. a gradual increase in the population at risk through reduced mortality and hence a greater benefit from a risk reduction), whereas the full life table method does account for this fact, as explained by Miller (2013).

Where life table methods are not applicable, e.g. due to data constraints, a simplified approach can be used instead (Miller et al. 2011). It uses a correlation based on national life expectancy to approximate YOLL due to changes in ambient $PM_{2.5}$ concentration levels, cf. equation 6.1.

$$\text{YOLL per 100 000 people (aged 30+)} = \exp\left(8.161 - (0.04478 \times \text{life expectancy})\right) \tag{6.1}$$

This simplified approach, also recommended by Holland (2014b), is used to derive a European average YOLL factor, applied to all countries except France in the European assessment domain (cf. Table A.5). It relies on life expectancy data for 2009 provided by the WHO (2016). No differentiation is made between males and females, having, however only a minor influence on the YOLL estimates.

As an approximation and due to the lack of alternative available quantification approaches, the same European average YOLL factor is equally applied to estimate long-term exposure NO_2 mortality risks in a sensitivity analysis (cf. section 7.2.1).

6.1.4 Monetary valuation

Human health effects are valued by considering the following components, each requiring dedicated valuation methods (Markandya and Ortiz 2011):

- resource costs, e.g. treatment costs;
- opportunity costs, e.g. foregone income;
- disutility costs, e.g. related to suffering as a consequence of an illness.

Resource costs are typically valued using prices of marketable goods or services. Opportunity costs, e.g. concerning lost productivity, are estimated by relying on (labour) market data. For the valuation of non-market goods, e.g. in the case of disutility costs, specific methods are used. These use surrogate markets to elicit individual preferences either directly (stated preferences) or indirectly (revealed preferences).

All three components should be valued for each health effect when following the theoretical concept of total economic value (Boardman et al. 2006), whilst leaving aside non-use values. In practice, this can be achieved, for instance, by using specifically designed stated preference methods, such as choice modelling or the contingent valuation method (CVM). A complementary, more pragmatic approach is to capture these different components through different endpoints, e.g. separating resource, opportunity, and disutility costs. This is what has been regularly done in European impact assessments.

Table 6.8 displays the monetary values per health endpoint, largely following latest European recommendations (Holland 2014a), that are used within the health damage cost assessment framework. As can be seen from the table, the three valuation components are covered through different endpoints and double counting is largely[22] avoided through using separate endpoints for related effects.

6.1.4.1 Monetary valuation of mortality risks

The monetary valuation of mortality risks has a predominant influence on quantifiable health damage costs and, given its intangible nature, involves higher uncertainties than the valuation of morbidity. In order to confront ethical concerns, it is important to stress that related assessments value a risk reduction of dying prematurely due to air pollution and not human life as such.

The reference VOLYs proposed in Table 6.8 and used in chapter 7 differ from the core values that have been used to support the impact assessment of recently proposed European air quality policies (Amann et al. 2014, Holland 2014a) and that in turn rely on a survey carried out in three countries during a previous European research project (European Commission 2004, Hurley et al. 2005). Newer evidence, however, is available from the NEEDS project (Desaigues et al. 2011), i.e. a study conducted in nine European countries and that focuses specifically on air pollution-related mortality risks. It uses improved stated preference techniques and is therefore preferred for the given setting (cf. also the parameter choices in chapter 4).

Mortality valuation remains a debated issue and alternative valuation approaches exist, e.g. using the value of a statistical life (VSL) approach, such as preferred in the US context (U.S. EPA 2012c) or other international institutions (OECD 2012), but also used for the valuation of infant mortality hereafter. Using a VSL-based approach is therefore tested in a sensitivity analysis (cf. section 7.2.2).

6.1.4.2 Adjustment of monetary valuation parameters across space and time

When using existing reference values for the assessment of environmental or health impacts in a different setting, i.e. concerning space or time, several adjustments should be considered (Pearce et al. 2006):

[22] One exception can be noted: In valuing net restricted activity days (RAD), it is not entirely consistent to include a component that refers to lost productivity, which relates to work loss days that are quantified separately. However, for consistency reasons and due to the minor importance of net RAD in the overall results, the valuation as suggested by Holland (2014a) will nevertheless be used.

- Time value of money, i.e. the use of discounting, cf. section 6.3;

- Price level changes (inflation): Given that the values in Table 6.8 were originally provided for the monetary base year 2005, an annual average inflation rate of 1.87% within the Euro zone of for the period 2005 to 2015 has been used to adapt the values to the monetary base year 2015 (eurostat 2016b). On this basis, an inflation adaption factor of 1.2 is obtained;

- Individual income changes and impact on willingness-to-pay: When transferring health-related values (especially intangible values derived by willingness-to-pay surveys) over time, it is common practice to adapt these for changes in individual income of the concerned population. Following OECD (2012) recommendations, the assumptions below are used to derive an adaptation factor:

    - Growth in individual income: real Gross Domestic Product (GDP) per capita in the EU28: 0.64% per year on average between 2005 and 2015 (eurostat 2016c);

    - Elasticity of willingness-to-pay with regard to personal income: assumed to be 0.8, following OECD (2012).

On this basis, an upscaling factor for the period from 2005 to 2015 of about 5.2% is obtained (((1+0.0064)^0.8)^10=1.0524).[23] This upscaling factor is applied in all calculations presented in chapter 7.

[23] As a simplification, it is assumed that all original health valuation factors used here were initially representing the willingness-to-pay of the year 2005.

Table 6.8: Default monetary valuation factors (net of income growth) and included components used in the assessment framework developed in this thesis (own compilation)

Endpoint	Value [$€_{2015}$][a]	Reference	Resource costs	Opportunity costs	Disutility costs
all-cause infant mortality (VSL)	*1925675*	Adapted from European Commission (2004), Hurley et al. (2005)		Via CVM	Via CVM
all-cause natural mortality (VOLY, long-term exposure)	*48142*	(Desaigues et al. 2011)		Via CVM	Via CVM
all-cause natural mortality (VOLY, acute exposure)	*72213*	Based on Desaigues et al. (2011)[b]		Via CVM	Via CVM
asthma symptom day	*51*	CVM based on Ready et al. (2004)			Via CVM
bronchitis prevalence/ symptom days	*708*	(Holland 2014a)		Based on asthma symptom day (14 days à 47 €)	Based on asthma symptom day
chronic bronchitis case	*64510*	(Holland 2014a), based on Hunt et al. (2011) and CVM by Maca et al. (2013)[c]	Via CVM	Via CVM	Via CVM
hospital admission (cardiovascular/ respiratory)	*2672*	(Hurley et al. 2005), CVM based on Ready et al. (2004)	Hospitalisation costs (45%)	Lost productivity (33%)	Via CVM (22%)
minor restricted activity day (MRAD)	*51*	CVM based on Ready et al. (2004)			Via CVM
(net) restricted activity day (RAD)	*111*	(Hurley et al. 2005), based on CBI 1998 study		Lost productivity (direct costs)	Via CVM

Endpoint	Value [$€_{2015}$][a]	Reference	Resource costs	Opportunity costs	Disutility costs
work loss day (WLD)	*156*	(Holland 2014a), based on CBI 2013 study		Lost productivity (direct costs)	

[a] Parameters in $€_{2000}$ or $€_{2005}$ have been converted to $€_{2015}$ using the above mentioned assumptions on price level changes; an adjustment for income growth has not (yet) been carried out for the numbers presented here

[b] Assuming that the original value given by Desaigues et al. (2011) was implicitly discounted, the valuation of acute effects is up-scaled by a factor of 1/0.67, following European Commission (2005a)

[c] Presumably, this value is a weighted average of the valuation of chronic bronchitis, mild chronic obstructive pulmonary disease (COPD) and severe COPD

6.2 Costing methodology

This section describes how the private costs for emission control techniques are characterised and provides specific values for use in the case study of chapter 7.3.1. Typical reference values and power plant data are used to derive annual costs over the power plant's life time and to display marginal abatement costs (per tonne of emission reduced) over a range of full-load hours per year.

6.2.1 Approach taken for techno-economic cost assessment

The approach used here is based on the Task Force on Techno Economic Issues[24] (TFTEI 2015), working under the mandate of the United Nations Economic Commission for Europe (UNECE) Convention on Long-Range Transboundary Air Pollution (CLRTAP). The TFTEI methodology builds on U.S. EPA (2002), while adapting elements and data for use in the European context and referring also to the EU's Reference Document on Economics and Cross Media Effects (European Commission 2006a). Among publicly available data, it is therefore considered a robust choice for cost estimation at plant level, striking a good balance between comprehensiveness and a manageable level of technical detail.

[24] Previously called Expert Group on Techno-Economic Issues (EGTEI); The spreadsheet is available on the TFTEI website: http://tftei.citepa.org/work-in-progress/costs-of-reduction-techniques-for-lcp, last accessed: 2017-05-18

To obtain the total annual costs, operational expenditures are added to the annualised capital costs (cf. section 3.3). Three specific, typical emission control measures are analysed in the following, i.e.:

- A dry low-NO_x burner (LNB) for NO_x emission reduction,
- switching to heavy fuel oil with reduced sulphur contents for SO_2 reduction, and
- an electrostatic precipitator (ESP) for PM reduction.

Reduction measures can be generally divided into primary and secondary measures. Primary measures aim at directly reducing the environmental burden at its source, e.g. during the combustion process. Secondary measures are also called end-of-pipe measures since they are designed to reduce emissions after their formation.

Note that even though the TFTEI spreadsheet allows considering part-load operation, this feature requires that efficiency losses and related emissions are determined by the user. Given that this data has not been available for the case studied here, the cost characterisation is based on full-load operation equivalents. Real reduction costs are therefore expected to be somewhat higher than those presented below.

6.2.2 Cost characterisation of potential emission control measures at a heavy fuel oil-fired power plant

The costs of retrofitting specific emission control measures at existing heavy fuel oil-fired power plant units are estimated using a set of general parameter assumptions (Table 6.9) and the default settings of the TFTEI spreadsheets. A simple script was developed to generate cost estimates over a range of full load hours that are displayed below.

Operating devices such as LNB and ESP not only leads to additional electricity consumption, reflected in the operating costs, but also to slightly increasing atmospheric emissions per unit of electricity produced. Due to the minor importance, this effect on emissions has been disregarded in the cost characterisation.

Table 6.9: General parameters used for cost characterisation of emission control measures (own assumptions)

Parameter	Value
Monetary base year	$Euro_{2015}$
Thermal capacity per power plant unit	1715 MW_{th}
Electric efficiency	35%
Heavy fuel oil lower heating value at 0.33% sulphur	40.55 MJ/kg (based on detailed fuel composition)
Plant lifetime (= equipment lifetime)	20 years
Interest rate (weighted average cost of capital)	6% (standard scenario)
Electricity costs	40 € per MWh_{el}
Fixed operation & maintenance (O&M) costs	3% of investment
Full load hours	500 full load hour equivalents per year, corresponding to a capacity factor of 5.7% (standard scenario)

6.2.2.1 Cost characterisation of low NO_x burner (LNB)

Installing a LNB is a primary measure (i.e. modification of the combustion process) to reduce NO_x emissions. The injection of fuel and the air/oxygen mix into the combustion zone is altered, enabling to reduce the formation of both fuel NO_x and thermal NO_x (European Commission 2006b).

For the case studied here, the following specific assumptions are taken:

- NO_x concentration reduced from 600 to 400 mg/Nm³ through installing a LNB;
- specific equipment investment of 5.7 $€_{2015}/kW_{th}$.

Per power plant unit, these assumptions yield (TFTEI 2015):

- capital costs of 9 775 500 $€_{2015}$ (= 5.7 $€_{2015}/kW_{th}$ * 1715000 kW_{th});
- annual operating and maintenance costs of 293 265 $€_{2015}$ (3% of 9 775 500 $€_{2015}$).

Applying the annuity method and using equation 6.1 this translates into annualised costs of 1 145 538 $€_{2015}$, or 7 261 $€_{2015}$ per tonne of NO_x reduced (at 500 full load hour equivalents).

Given the fixed percentage of operation and maintenance costs as well as a specific investment that depends only on the power plant capacity, the annualised costs are independent from actual full load hours. This represents a simplification in that a more intensive use potentially leads to an earlier replacement of technical components, not being considered here.

Annualised costs per tonne of NO_x reduced are dependent on full load hours, illustrated by the graph below (Figure 6.8). It shows a remarkable cost per tonne increase at low utilisation rates (i.e. low operating hours), mainly driven by annualised capital costs.

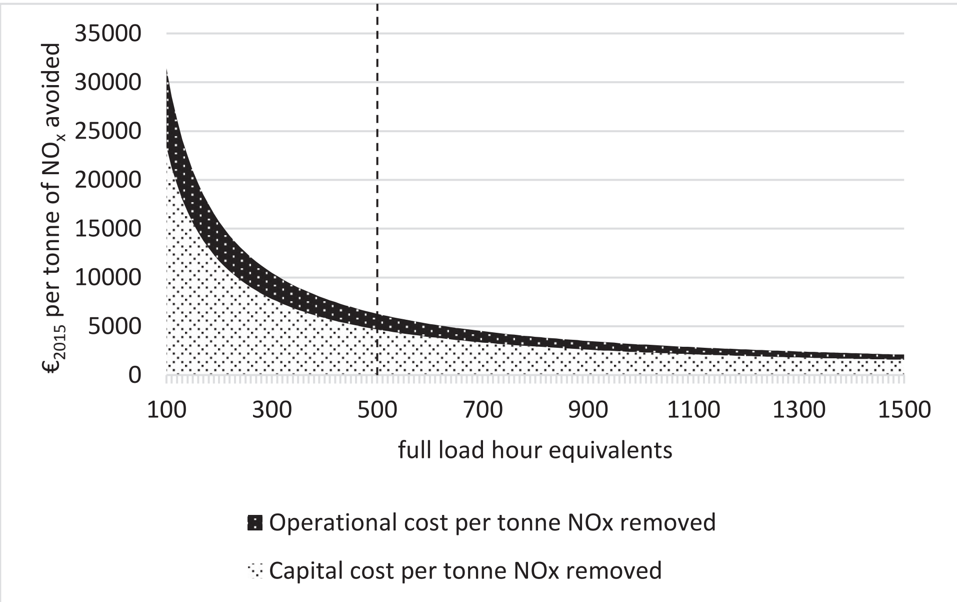


Figure 6.8: Cost curve (quasi-marginal abatement costs) for installing a low NO_x burner at an oil-fired power plant unit; dotted line at 500 h/a; based on own calculations

6.2.2.2 Fuel switching to very low/ultra-low sulphur heavy fuel oil

A relatively simple and low-cost primary measure for SO_2 emission reductions is fuel switching towards heavy fuel oil with lower sulphur contents. A potential constraint is the availability of heavy fuel oil with ultra-low sulphur contents, but new European regulations requiring lower sulphur contents in the shipping sector (i.e. Directive 2012/33/EU) are likely to increase the supply of such fuels.

For the case studied here, it is assumed that the use of heavy fuel oil with 0.33% sulphur content allows to achieve a SO_2 flue gas concentration of 600 mg/Nm³ and reducing to 0.22% sulphur allows to respect 400 mg/Nm³.

The price differential for moving from a fuel with 0.5% sulphur content to 0.1% sulphur content is around 199-229 $€_{2009}$/tonne. This value is based on Kalli et al. (2009) and applies to shipping fuels. Since a literature search for data applying to combustion fuels did not deliver any results, this value is adopted for the case studied here. Assuming an average Euro zone inflation rate of 1.47% between 2009 and 2015 (eurostat 2016b), the lower bound translates into 54.30 $€_{2015}$ per tonne per 0.1% sulphur content change. The specific reduction costs per tonne of emission reduced are constant, as they increase linearly with full load hours and thus with emission quantities. For these parameters choices, the abatement costs for SO_2 when switching from fuel with 0.33% sulphur to 0.22% sulphur amount to approximately 26 146 $€_{2015}$ per tonne of SO_2 removed (TFTEI 2015).

In order to estimate annual costs for this control measure, the total amount of reduced SO_2 per year and dependent on operating hours needs to be defined. Assuming a concentration decrease from 600 mg/Nm³ to 400 mg/Nm³, a flue gas volume stream of 1 800 000 Nm³/h (Table 6.2) and 500 full load hours per year, this yields an annual reduction of 180 tonnes[25] of SO_2. Multiplying these with the marginal cost of 26 146 $€_{2015}$ per tonne of SO_2 removed therefore leads to annual costs of 4 706 280 $€_{2015}$ for the fuel switching per power plant unit and the given operating time.

6.2.2.3 Cost characterisation of an electrostatic precipitator (ESP)

For dust emission reduction, electrostatic precipitators are the most commonly applied secondary reduction technique in large combustion plants. The functional principle is to apply an electric charge to gaseous dust particles within the flue gas stream that are subsequently absorbed at oppositely charged collector plates (European Commission 2006b).

[25] (600-400) mg/Nm³ * 1 800 000 Nm³/h * 500 h/a * (1 t / 10^9 mg)

For the case studied here, the following specific assumptions are taken:

- installing an ESP allows achieving a dust concentration of 20 mg/Nm³;
- disposal of waste (fly ash);
- ESP material is carbon steel.

Combining the above with further standard assumptions (e.g. 500 full load hour equivalents), the total annualised costs amount to 471 664 €$_{2015}$ per unit or 32 142 €$_{2015}$ per tonne of PM reduced (TFTEI 2015).

The following graph (Figure 6.9) displays the specific costs per tonne of PM reduced over a range from 100 to 1500 full load hours per year, illustrating, once again, a steep cost increase at low utilisation rates.

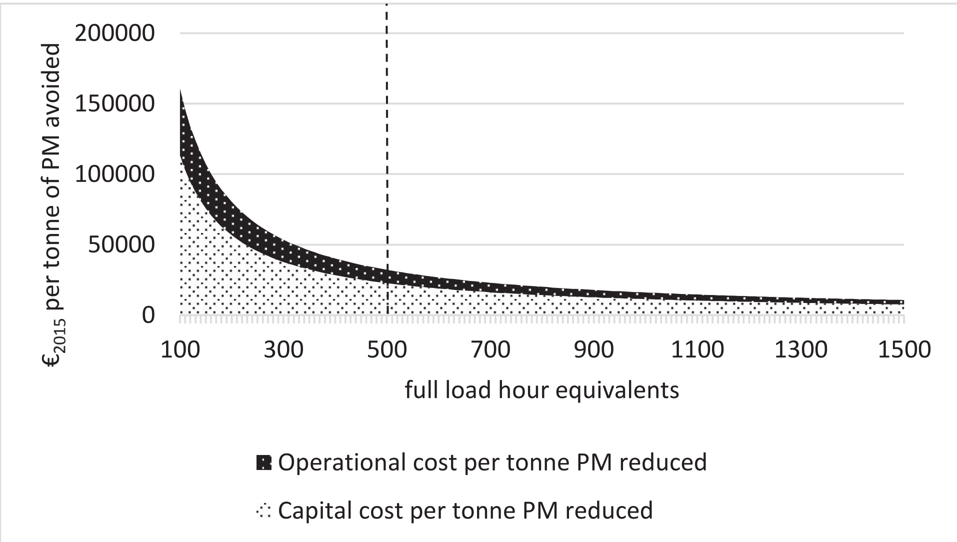


Figure 6.9: Cost curve (quasi-marginal abatement costs) for installing an electrostatic precipitator at an oil-fired power plant unit; dotted line at 500 h/a; based on own calculations

6.3 Discount rates used in case studies

In accordance with the scientific literature on discounting in CBA (e.g. Boardman et al. 2006, Harrison 2010, Pearce et al. 2006, Rabl et al. 2014) and the principles described in section 3.2.3, the following discount rates are adopted:

- Future annual costs and benefits are discounted at an equal social discount rate of 4% (European Commission 2009) given the European scope of the assessment.

- Using a social discount rate in the CBA is complementary to using a private interest rate for the private cost characterisation. Here, taking the perspective of a publicly owned utility, the weighted average cost of capital is assumed to be 6% by default. Due to the fact that this is rather at the low end of typical private interest rates in the energy sector (Capros et al. 2016), alternative assumptions are tested in a sensitivity analysis in section 7.3.3. The combination of a private interest rate and social discount rate is called the two-stage discounting procedure, further described by Kolb and Scheraga (1990).

6.4 Summary of chapter 6

The methodological framework for the assessment of health damages related to atmospheric emissions as well as the cost characterisation of emission control measures are described. The health damage cost assessment follows the stages of the impact pathway approach, i.e. definition of emission scenarios, exposure assessment, health impact assessment and monetary valuation. The complete modelling chain, its technical implementation, input data and related assumptions are transparently and comprehensively documented. Moreover, the emission scenarios and specific emission control costs used for the subsequent case studies are presented.

7 Application of the extended methodology framework and results

The newly developed methodology framework for the economic assessment of emission control measures at power plants with variable operation profiles is applied to several case studies. These are grouped according to their global objectives (Table 7.1).

The single cases are addressed through specific objectives that are respectively detailed using the following scheme: *What* (is the aim?), *Where* (i.e. at what scale are impacts assessed?), *For* (which emission scenarios?), and *Using* (which general and specific methodological assumptions?).

Table 7.1: Overview on research objectives and corresponding case studies treated in the current chapter

Research objectives and corresponding case studies	Section
Exploring the influence of emission patterns and atmospheric modelling features	
Emission intensity	7.1.1
Background emission inventory	7.1.2
Temporal modelling resolution	7.1.3
Spatial modelling resolution	7.1.4
Exploring the influence of methodological choices in health impact assessment	
Inclusion of NO_2-related health endpoints	7.2.1
Alternative long-term exposure mortality impact assessment	7.2.2
Lowering the concentration threshold for ozone-related health impacts	7.2.3
Social CBA of emission control measures	
Primary and secondary emission control measures	7.3.1
Comparison with simplified approaches for benefit assessment	7.3.2
Sensitivity analysis regarding the social CBA	7.3.3

To enable a consistent comparison of different power plant operation scenarios, the resulting health damage costs are expressed in € per MWh of electricity generated.

In order to assess the effects that can be attributed to a specific power plant operation scenario, the respective health impacts are expressed relative to a so-called baseline scenario, i.e. without atmospheric emissions from the power plant. This baseline scenario is labelled 'S0' (cf. also Table 6.2).

7.1 Exploring the influence of emission patterns and atmospheric modelling features

This section serves to analyse the influence of different emission patterns on health damage costs, i.e. in terms of emission intensity and annual distribution. At the same time, by varying the spatial modelling resolution and by simulating a different temporal modelling resolution (constant versus variable emission pattern), it also permits to study the influence of atmospheric modelling features on health damage costs. It should be noted that nesting, i.e. integrating results from a sub modelling domain (e.g. France) into the parent modelling domain (e.g. Europe), was not carried out for the results presented hereafter. Adding the results from different scales would thus lead to an overestimation of the total health damage costs.

7.1.1 Emission intensity

Table 7.2 summarises the objective and approach to analyse the influence of emission intensity, i.e. the quantity of a pollutant emitted per time step. As shown in Table 6.2, there is a large difference in the annual emission quantities per pollutant between both scenarios compared here (> factor of 17).

Table 7.2: Objective and research approach to assess the influence of emission intensity at source level on health damage costs

Objective	What is the influence on health damage costs of varying the emission intensity (emission quantity released per hour) at the source?
Approach	
What	Assessing the influence of emission intensity on the health damage costs related to $PM_{2.5}$, PM_{10} and O_3 by comparing a constant operation scenario with normal and high emission intensity respectively
Where	Île-de-France (IDF), France (FR), Europe (EU), cf. Figure 6.3 and Figure 6.4
For	Scenarios S2b (normal emission intensity) and S2c (high emission intensity), compared to S0 (baseline), cf. Table 6.2
Using	Standard modelling framework as described in section 6.1

As illustrated in Figure 7.1, an increase in emission intensity of a point source leads to a decrease in health damage costs (per unit of electricity generated). At the EU modelling domain, this reduction amounts to 3%, whereas it is more pronounced at the France domain (- 15%) and at the Île-de-France domain (- 27%).

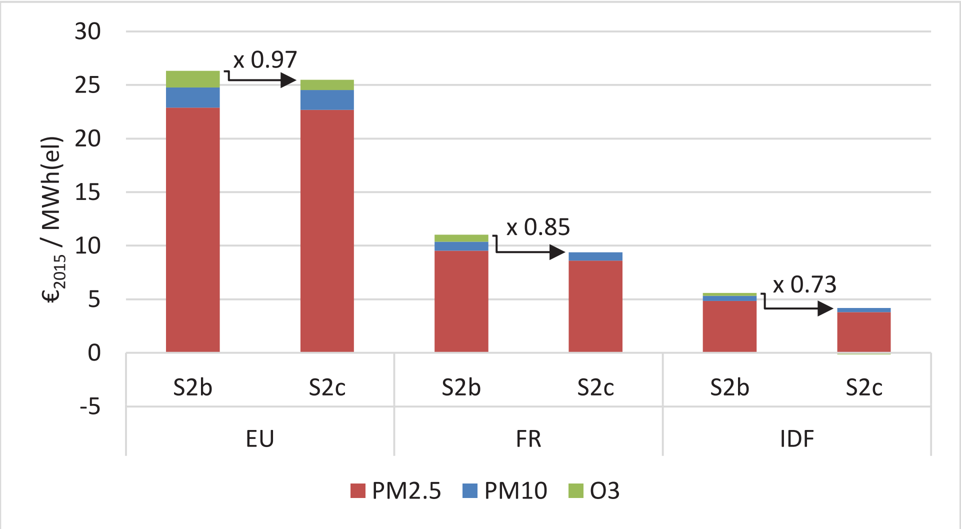


Figure 7.1: Annual health damage costs related to the normal emission intensity scenario (S2b) and the high emission intensity scenario (S2c) at different modelling domains

7.1.2 Background emission inventory

Table 7.3 summarises the objective and approach to analyse the influence of the background emission inventory on health damage costs. Such an inventory includes all natural and anthropogenic emissions considered during the atmospheric modelling. Globally, emissions of all relevant pollutants are expected to be lower in the future (year 2020) scenario than in the year 2009. The analysis was only conducted at the French modelling domain in order to limit the computational effort.

Table 7.3: Objective and research approach to assess the influence of the background emission inventory on health damage costs

Objective	What is the influence on health damage costs of using an assumed future background emission inventory as compared to the standard background emission inventory?
Approach	
What	Assessing the influence of the background emission scenario on the health damage costs related to $PM_{2.5}$, PM_{10}, and O_3 by comparing results for the standard (2009) and a hypothetical future (2020) background emission inventory
Where	France (FR), cf. Figure 6.3
For	Scenario S2c, compared to S0 (baseline), cf. Table 6.2
Using	Standard modelling framework as described in section 6.1 with a variation in the background emission inventory, cf. Table 6.4

As shown in Figure 7.2, changing the background emission scenario leads to a minor (+ 3%) increase in health damage costs at the France modelling domain.

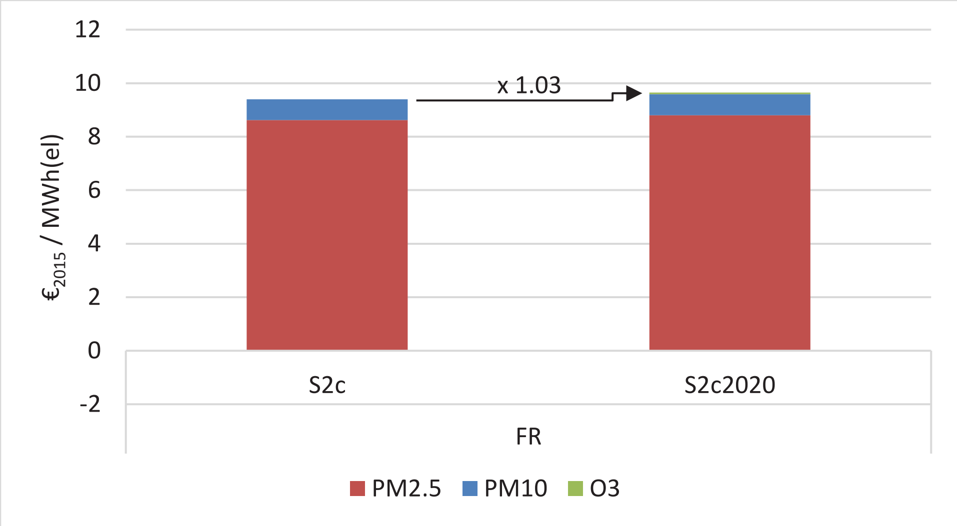


Figure 7.2: Annual health damage costs related to emission scenario S2c using the default background emission inventory (left) and a hypothetical future background emission inventory 2020 (right) at the France modelling domain

7.1.3 Temporal modelling resolution

Table 7.4 summarises the objective and approach to analyse the influence of the chosen temporal modelling resolution. In the given case, all atmospheric modelling results are calculated at an hourly basis. By evaluating a scenario with a constant hourly emission rate, the functioning of an averaged modelling approach that does not account for variations in emission patterns throughout the year is simulated. Such an average modelling is notably underlying the parameterised results of the EcoSenseWeb model (cf. section 5.5.2). This average modelling approach is compared to a more highly resolved modelling approach that accounts for variations in emissions throughout the year.

Table 7.4: Objective and research approach to assess the influence of temporal resolution on health damage costs

Objective	What is the influence on health damage costs of using a higher temporal modelling resolution, accounting for variable operation profiles of the emission source throughout the year?
Approach	
What	Assessing the influence of temporal modelling resolution on the health damage costs related to $PM_{2.5}$, PM_{10}, and O_3 by comparing a constant emission scenario to a variable emission scenario with equivalent overall emissions during one year
Where	Île-de-France (IDF), France (FR), Europe (EU), cf. Figure 6.3 and Figure 6.4
For	Scenarios S2 (variable operation) and S2b (constant operation), compared to S0 (baseline), cf. Table 6.2
Using	Standard modelling framework as described in section 6.1

Figure 7.3 shows the influence of using a different temporal resolution on health damage costs, all other input data being equal.

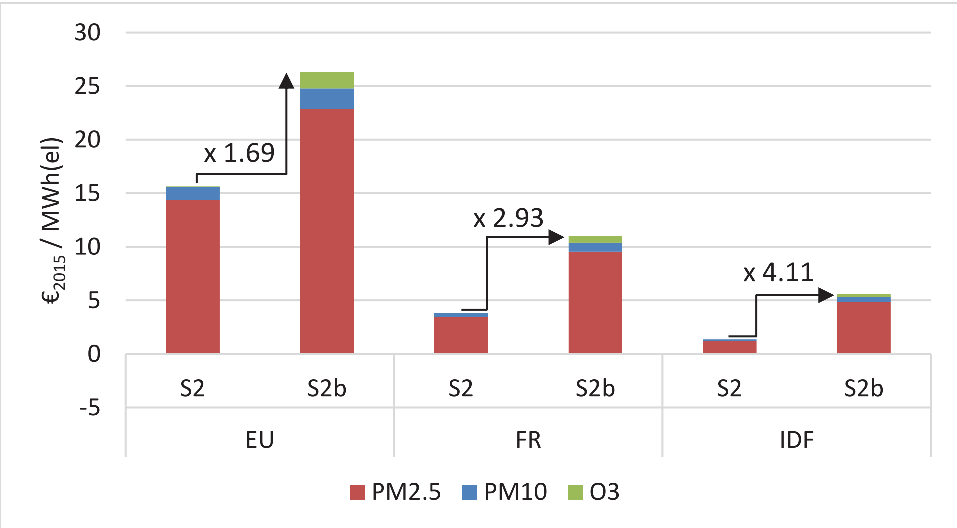


Figure 7.3: Annual health damage costs related to the variable emission scenario (S2) and the equivalent constant emission scenario (S2b) at different modelling domains

Compared to the variable emission scenario (S2), the constant emission scenario (S2b) leads to an increase in health damage costs by a factor of 1.69 at the European domain, whereas for the French (factor 2.93) and IDF domain (factor 4.11), the increase is even higher.

To better understand these differences, it is instructive to look at the concentration fields of annual mean $PM_{2.5}$ concentration, e.g. in France (Figure 7.4). Apart from a globally higher concentration increase in the constant emission scenario, the spatial distribution pattern is also different, reflecting further influences, for instance different wind directions throughout the year (Figure 7.5).

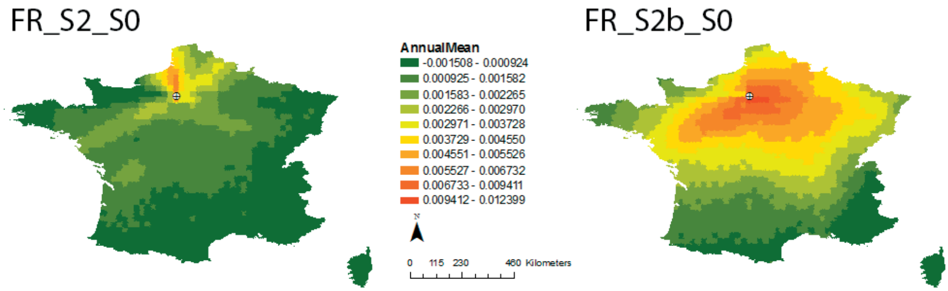


Figure 7.4: Change in annual ambient mean $PM_{2.5}$ concentration due to the variable operation scenario S2 (left) and the equivalent constant emission scenario S2b (right) at the France modelling domain

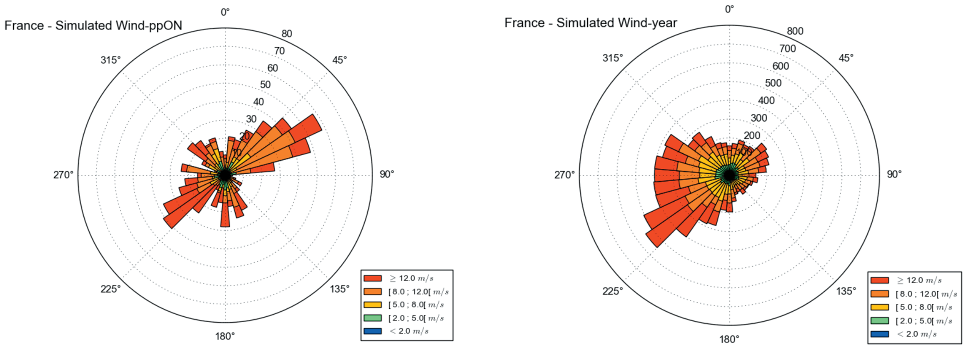


Figure 7.5: Wind directions and wind speed simulated using the WRF model at emission release height over the France modelling domain during the variable operation scenario (left) and during the constant operation scenario (right); Source: Legorgeu (2016)

7.1.4 Spatial modelling resolution

Table 7.5 summarises the objectives and related approaches to analyse the influence of spatial modelling resolution, i.e. the resolution of the grid cells for which results are estimated.

Table 7.5: Objectives and research approach to assess the influence of spatial resolution on health damage costs

Objec-tives	1) What is the influence on health damage costs of using a different spatial modelling resolution for a fixed spatial perimeter? 2) What is the influence on health damage costs of using a so-called plume-in-grid modelling approach, i.e. a local Gaussian modelling embedded into the less resolved Eulerian modelling?
Approach for objective 1)	
What	Assessing the influence of spatial modelling resolution on the health damage costs related to $PM_{2.5}$, PM_{10}, and O_3
Where	Île-de-France (IDF): high (~3 km x ~5 km), medium (~17 km x ~25 km) and low (~35 km x ~50 km) modelling resolution, cf. Table 6.4 and Figure 6.3 France (FR): medium (~17 km x ~25 km) and low (~35 km x ~50 km) modelling resolution, cf. Table 6.4 and Figure 6.3
For	Scenario S2, compared to S0 (baseline), cf. Table 6.2
Using	Standard modelling framework as described in section 6.1, adapted by using the age group fractions, baseline rates and long-term exposure mortality impact function of the European modelling domain at all domains and thus allowing for a consistent comparison
Approach for objective 2)	
What	Comparing health damage costs related to $PM_{2.5}$, PM_{10}, and O_3 based on a plume-in-grid modelling approach with those based on the standard modelling approach
Where	Île-de-France (IDF), France (FR), cf. Figure 6.3
For	Scenarios S1, S1PIG, S2 and S2PIG, compared to S0 (baseline), cf. Table 6.2
Using	Standard modelling framework as described in section 6.1

Figure 7.6 shows the influence of modelling grid resolution on health damage costs, all other elements being equal. For the modelling domain France, using a higher spatial resolution results in a 9% decrease in health damage costs. For the Île-de-France domain, the direction of change depends on the resolution: moving from a low to medium spatial resolution results in a 2% increase, whereas moving to a high modelling resolution results in a 3% decrease in health damage costs. Using a nesting approach at European level would thus slightly decrease the health damage costs compared to the standard approach without nesting.

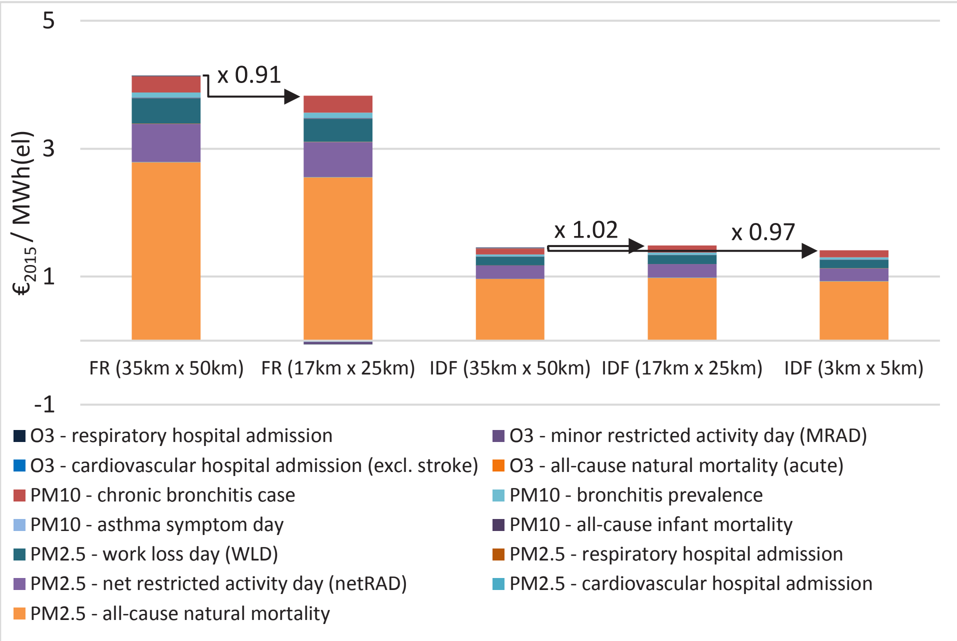


Figure 7.6: Annual health damage costs related to emission scenario S2 at different modelling domains and assessed using different spatial modelling resolutions

Regarding the second objective, Figure 7.7 shows the influence of the plume-in-grid (PIG) modelling on the results.

In all analysed cases, the PIG modelling leads to a considerable increase in health damage costs, ranging from a factor of 1.75 (France domain) to a factor of 2.51 (Île-de-France domain). At the local scale, the effect is more pronounced than at the national scale.

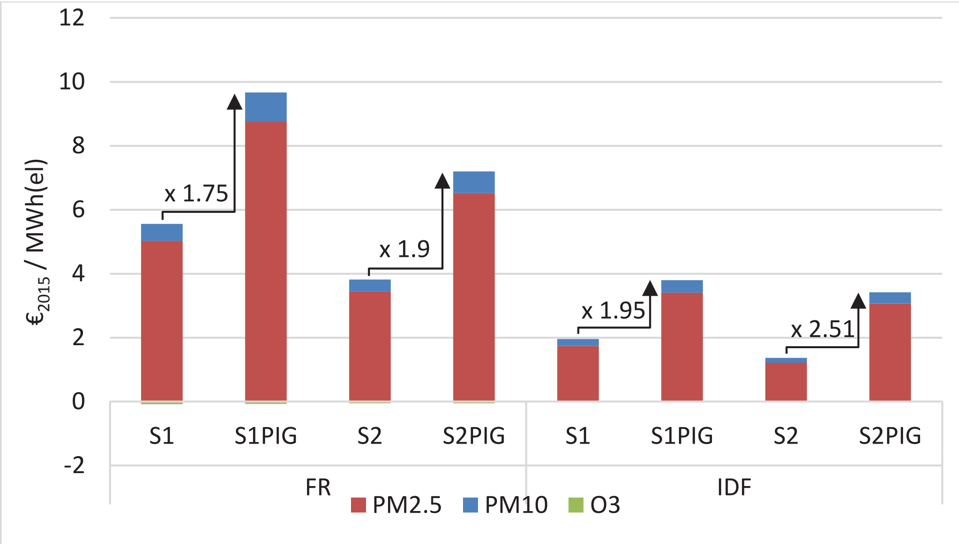


Figure 7.7: Health damage costs caused by emission scenarios S1 and S2 using the standard or the plume-in-grid (PIG) modelling approach at the France and Île-de-France modelling domain

To better understand these results, dominated by $PM_{2.5}$-related health damage costs, the $PM_{2.5}$ concentration changes at the local (IDF) domain are displayed using either a medium or high spatial resolution (Figure 7.8) and for both types of atmospheric modelling (i.e. with or without PIG, Figure 7.9). The colour ranges have been harmonised in order to allow for direct comparisons within each figure, however not across the two figures.

For the IDF modelling domain and the area around the emission source (highlighted by a white dot), a medium spatial modelling resolution leads to a slightly higher level of $PM_{2.5}$ concentrations than the higher modelling resolution (Figure 7.8), partly explaining the small difference in health damage costs observed above.

Using the plume-in-grid modelling approach leads to a different pattern of concentration changes around the emission source as well as to remarkably higher concentration increases compared to the standard modelling approach (Figure 7.9). This explains the increases in health damage costs for the PIG modelling observed above.

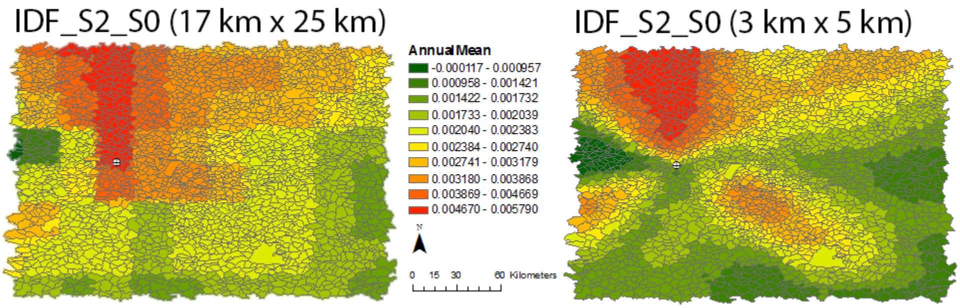


Figure 7.8: Change in annual ambient mean $PM_{2.5}$ concentration due to operation scenario S2 at the Île-de-France modelling domain using a medium (left) or high (right) spatial modelling resolution

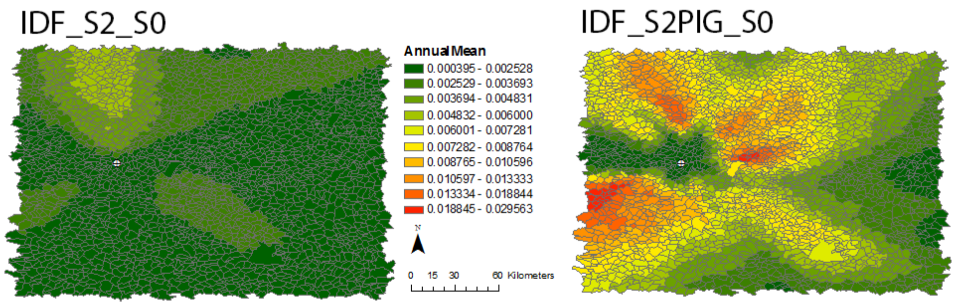


Figure 7.9: Change in annual ambient mean $PM_{2.5}$ concentration due to the variable operation scenario S2 using the standard (left) or plume-in-grid (PIG) modelling approach (right) at the Île-de-France modelling domain

7.2 Exploring the influence of methodological choices in health impact assessment

Apart from methodological choices in atmospheric modelling, the input data and parameter choices in health impact assessment have already been shown to substantially influence quantifiable health damage costs (cf. chapter 4). Complementary to these findings and following latest recommendations by the WHO (2013a), some additional elements shall be analysed here. Most importantly in view of recent findings, the influence of NO_2-related health impacts on overall quantifiable health damage costs is studied. Given the predominant role of mortality-related health damage costs, particularly related to long-term exposure to $PM_{2.5}$, an alternative quantification approach for this endpoint is tested. This is followed by a case study in which the concentration threshold for ozone-related health impacts is varied.

7.2.1 Inclusion of NO_2-related health impacts

Table 7.6 summarises the objective and approach to analyse the influence of additionally quantifying NO_2-related health impacts. The scenarios S2 and S2b were chosen for illustrative purposes. The influence of including NO_2-related health impacts in social CBA is assessed in section 7.3.3.

Table 7.6: Objective and research approach to assess the influence of additionally quantifying direct NO_2-related health effects

Objective	What is the effect on health damage costs of quantifying direct NO_2-related endpoints in addition to PM and ozone-related endpoints?
Approach	
What	Including direct NO_2-related endpoints in the health damage cost assessment
Where	Île-de-France (IDF), France (FR), Europe (EU), cf. Figure 6.3 and Figure 6.4
For	Scenarios S2 (variable operation) and S2b (constant operation), compared to S0 (baseline), cf. Table 6.2
Using	Standard modelling framework as described in section 6.1 and additionally accounting for NO_2-related health endpoints (Table 6.7), whilst multiplying the long-term NO_2 exposure-related mortality impact function with a factor of 0.67 in order to account for potential double counting risks with regard to PM (WHO 2013a)

Figure 7.10 shows the absolute contribution of NO_2-related health damage costs at the three modelling domains under different operation scenarios. In addition, the relative contribution of the only substantial NO_2-related endpoint, i.e. long-term NO_2 exposure-related mortality risks (above an annual threshold of 20 µg/Nm³), is displayed. NO_2-related health damage costs vary considerably across the modelling domains. Whilst their contribution amounts to only a few percent at European level, it rises to at least 14% at the French level and at least 25% at the Île-de-France level, depending on the emission scenario.

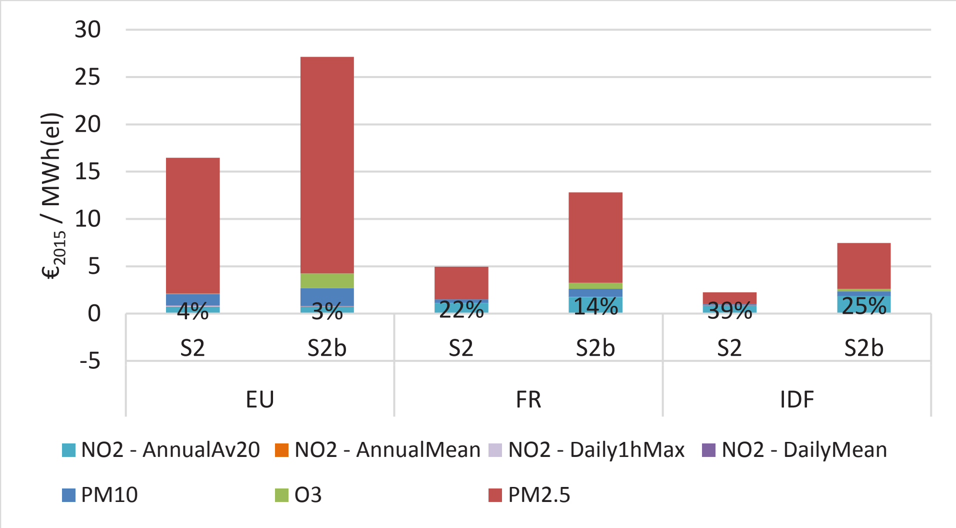


Figure 7.10: Annual health damage costs related to emission scenarios S2 (variable) and S2b (constant) showing the percentage contribution of NO_2-related mortality impacts due to long-term exposure (annual average above 20 µg/m³) at different modelling domains

Interestingly, the absolute level of NO_2-related health damage costs is approximately equal at Île-de-France level and France level and lower at European level. This can be mainly explained by the fact that the largest share of NO_2-related impacts occurs only above an annual ambient concentration threshold of 20 µg/m³. As shown in Figure 7.11, this threshold is only exceeded in the surroundings of the metropolitan area eastwards of the emission source. This means that a relatively small area is responsible for the quantified health damage costs at national level, explaining the quasi-parity of health damage costs at the local and national level. At the European scale, the ambient concentration increases at a lower absolute level (Figure 7.12), which can partly be explained by the coarser modelling resolution. Concentrations are thus diluted over a larger area. Even though some additional exposure occurs in the Benelux area and south-eastwards of France, the quantified health damage costs are lower overall than at the national level alone.

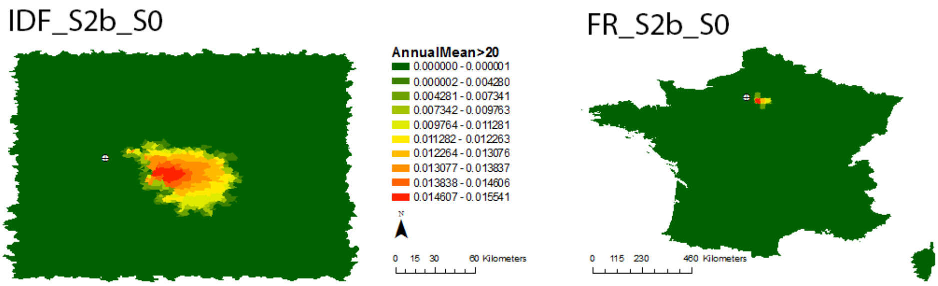


Figure 7.11: Change in annual ambient mean NO_2 concentration (above 20 µg/m³) due to operation scenario S2b at the Île-de-France (left) and France (right) modelling domain

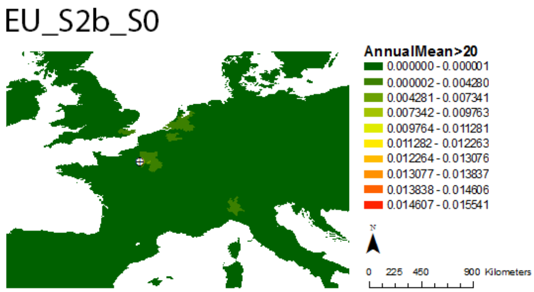


Figure 7.12: Change in annual ambient mean NO_2 concentration (above 20 µg/m³) due to operation scenario S2b at the European modelling domain (excerpt of modelling domain)

7.2.2 Using an alternative approach for long-term exposure mortality impact assessment

As shown in chapter 4, the approach for assessing long-term $PM_{2.5}$ exposure-related mortality impacts has a crucial influence, given the large share of damage costs that are regularly attributed to this single endpoint. As an alternative to quantifying years of life lost due to an increased mortality risk, the number of premature deaths are estimated here, following key international institutions (OECD 2012, U.S. EPA 2012c). Subsequently, these premature deaths are valued using the so-called VSL (Value of a Statistical Life, cf. also section 6.1.3.2). The analysis uses the following input parameters:

- The impact function is composed of the relative risk as recommended by WHO (2013a) and French and EU mortality baseline rates as given in Table A.4 (Appendix);
- For monetary valuation, the same VSL parameter as for all-cause infant mortality is used for all-cause mortality from long-term exposure, cf. Table 6.8, i.e. amounting to 1.9 million $€_{2015}$ per premature death, net of income adaption.

Table 7.7 summarises the objective and approach to analyse the influence of the approach for long-term exposure mortality impact assessment. In section 7.3.3, the effect of using an alternative approach in an exemplary social CBA is assessed.

Table 7.7: Objective and research approach to assess the influence of different approaches for long-term exposure mortality impact assessment

Objective	What is the impact of using an alternative approach to long-term $PM_{2.5}$ exposure mortality impact assessment?
Approach	
What	Assessing the influence of using the VSL approach instead of the VOLY approach for long-term exposure mortality impact assessment on health damage costs
Where	Île-de-France (IDF), France (FR), Europe (EU), cf. Figure 6.3 and Figure 6.4
For	Scenarios S2 (variable operation) and S2b (constant operation), compared to S0 (baseline), cf. Table 6.2
Using	Standard modelling framework as described in section 6.1, however using an alternative approach for long-term exposure mortality impact assessment and valuation, described above

Figure 7.13 shows the result of applying the alternative approach for mortality risk assessment. Depending on the modelling domain (and underlying different baseline mortality rates used), the VSL-based approach leads to an increase by a factor of 3.69 (European domain) and 3.09 (France and Île-de-France domain) respectively. The difference between the domains can be explained by the different mortality impact assessment approaches used (cf. section 6.1.3.2).

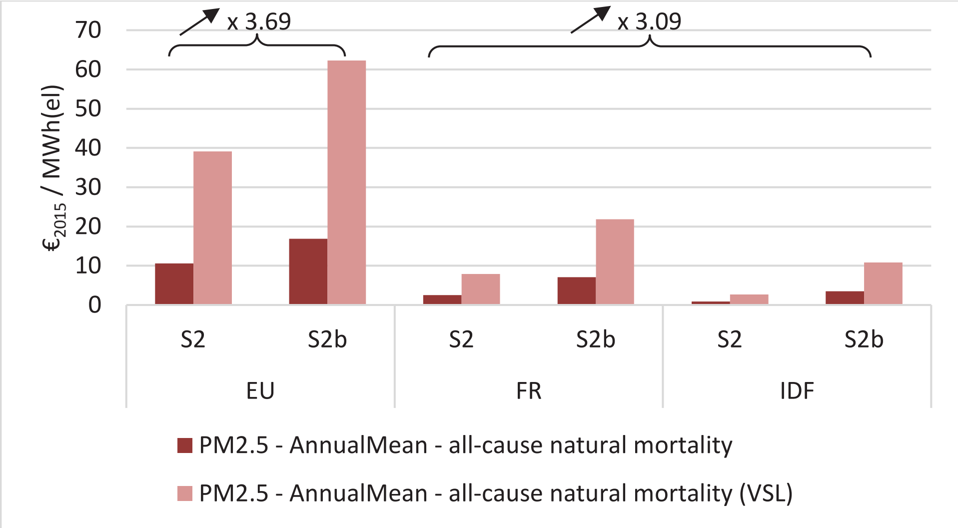


Figure 7.13: Annual mortality-related health damage costs due to long-term $PM_{2.5}$ exposure of the emission scenarios S2 (variable) and S2b (constant) at different modelling domains and estimated using two alternative approaches

7.2.3 Lowering the concentration threshold for ozone-related health impacts

Instead of the standard metric for ozone impacts, i.e. SOMO35 (Sum Of (maximum daily 8h) Means Over 35 parts per billion), the alternative metric SOMO10 (Sum Of (maximum daily 8h) Means Over 10 parts per billion) is used, as recommended by the WHO (2013a) for sensitivity analyses.

Table 7.8 summarises the objective and approach to analyse the influence of a lower concentration threshold for ozone-related health effects. The scenarios were chosen for illustrative purposes, i.e. reflecting different operation patterns and different emission intensities.

Table 7.8: Objective and research approach to assess the influence of different concentration threshold levels for ozone-related health effects

Objective	What is the effect of assuming a lower effect threshold for assessing ozone-related health damages (SOMO10 instead of SOMO35)?
Approach	
What	Assessing the influence of the concentration threshold above which ozone-related health effects are assumed to occur on ozone-related health damage costs
Where	Île-de-France (IDF), France (FR), Europe (EU), cf. Figure 6.3 and Figure 6.4
For	Scenarios S2, S2b and S2c, compared to S0 (baseline), cf. Table 6.2
Using	Standard modelling framework as described in section 6.1, however considering ozone effects above a concentration threshold of 10 ppb (instead of 35 ppb)

Figure 7.14 shows the influence of the alternative ozone metric on related quantified health damage costs.

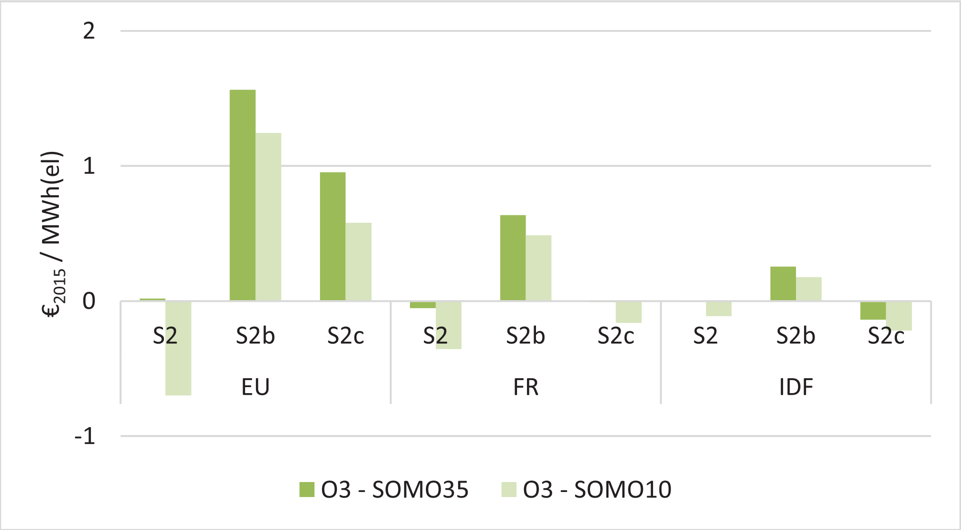


Figure 7.14: Annual ozone-related health damage costs for emission scenarios S2 (variable), S2b (constant, normal emission intensity) and S2c (constant, high emission intensity) at different modelling domains and using two alternative concentration thresholds

Interestingly, assuming a lower concentration threshold leads to lower health damage costs for all scenarios and at all scales. This somewhat paradox outcome cannot be fully explained, as it depends on the non-linear processes underlying tropospheric ozone formation that are dependent on the emission source, its location and background conditions during the year (cf. section 2.4.3).

A closer look at the results reveals further particularities. For instance, different emission intensities lead to either positive (S2b) or negative (= avoided) health damage costs (S2c) at the IDF and FR modelling domain, whereas they lead to positive health damage costs for both scenarios at the EU modelling domain. For the variable emission scenario (S2) and at the EU modelling domain, switching from SOMO35 to SOMO10 induces a change from positive health damage costs to negative health damage costs.

7.3 Social cost-benefit analysis of emission control measures

The health damage cost assessment framework (section 6.1) is combined with the costing methodology (section 6.2) in order to analyse the efficiency of installing selected primary and secondary emission control measures at the exemplary power plant using social CBA. The objective is to demonstrate the feasibility and methodological particularities when carrying out social CBA at a specific industrial site.

7.3.1 Social CBA of primary and secondary emission control measures

Table 7.9 summarises the objectives and related approaches in order to conduct social CBAs of emission control measures.

Table 7.9: Objectives and research approach to assess the private costs and health-related benefits of primary and secondary emission control measures

Objectives	1)	What are the private costs and health-related benefits of reducing NO_x and SO_2 emissions at a heavy fuel oil-fired power plant and are these measures socially efficient?
	2)	What are the private costs and health-related benefits of additionally reducing PM emissions and is this measure socially efficient?

Approach for objective 1)	
What	Assessing the health benefits (reduced damage costs related to $PM_{2.5}$, PM_{10}, and O_3) and associated private costs of primary NO_x and SO_2 emission control measures
Where	France (FR), Europe (EU), cf. Figure 6.3 and Figure 6.4
For	Scenarios S1 (initial emission level) and S2 (reduced emission level), compared to S0 (baseline), cf. Table 6.2
Using	Standard modelling framework as described in section 6.1 and private cost assessment described in sections 6.2.2.1 and 6.2.2.2
Approach for objective 2)	
What	Assessing the health benefits (reduced damage costs related to $PM_{2.5}$, PM_{10}, and O_3) and associated private costs of a secondary PM emission control measure
Where	France (FR), Europe (EU), cf. Figure 6.3 and Figure 6.4
For	Scenarios S2 (initial emission level, including primary NO_x and SO_2 emission control measures) and S3 (reduced emission level), compared to S0 (baseline), cf. Table 6.2
Using	Standard modelling framework as described in section 6.1 and private cost assessment described in section 6.2.2.3

The following parameter choices are made for the social CBAs carried out here:

- Benefits consist of avoided health damage costs due to emission control measures at the power plant;
- Costs consist of annualised investment and operating costs of the respective emission control measures (cf. section 6.2.2), estimated using a 6% weighted average cost of capital by default; an equipment lifetime of 20 years and 500 operating hours per year are assumed (cf. Table 6.9);
- Discount rate: Both future costs and benefits are discounted using a social discount rate of 4% (cf. section 6.3);
- Affected individuals: Given that the benefits of emission reductions occur within and beyond the French border, the analysis considers the population of the European modelling domain. In order to estimate the share of effects at national level, results for France are given in addition.

The health benefits of installing primary NO_x and SO_2 emission control measures, i.e. a low-NO_x burner and switching to heavy fuel oil with lower sulphur contents, can be derived from Figure 7.15.

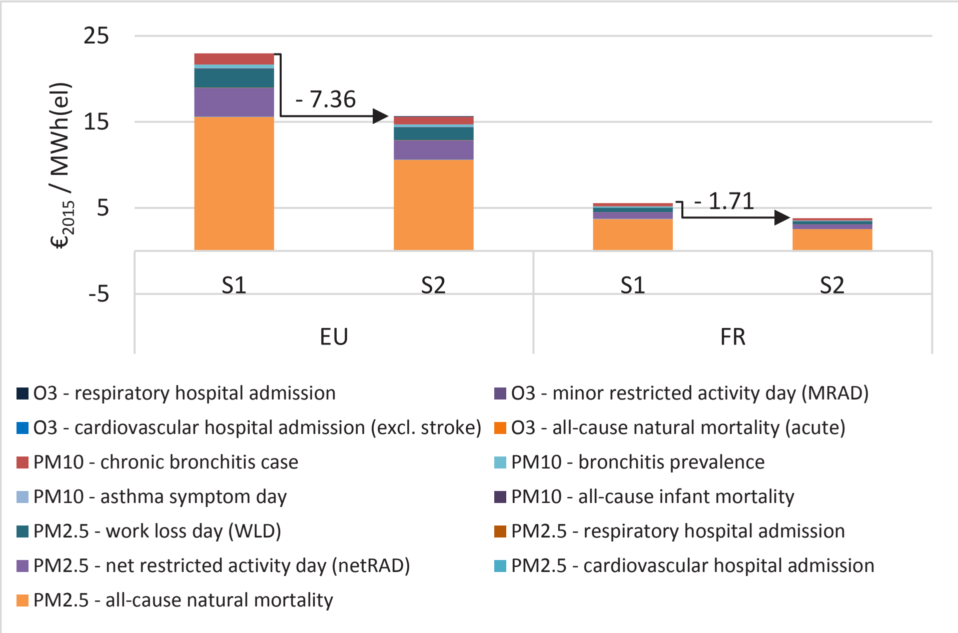


Figure 7.15: Annual health damage costs related to the initial emission level (S1) and after installing primary emission control measures for NO_x and SO_2 (S2) at different modelling domains

Moving from emission scenario S1 to emission scenario S2 reduces health damage costs by 7.36 $€_{2015}$ per MWh_{el} (Europe) and 1.71 $€_{2015}$ per MWh_{el} (France). When multiplied with the annual electricity generation, i.e. 300 000 MWh per power plant unit, this leads to annual benefits of 300 000 * 1.71 = 513 000 $€_{2015}$ (France) and 300 000 * 7.36 = 2 206 788 $€_{2015}$ (Europe). Hence, approximately one fourth of benefits occur at the national scale.

The annualised private costs of the envisaged emission reduction measures amount to (cf. section 6.2.2):

- 1 145 538 $€_{2015}$ per year and unit for installing the low NO_x burner;
- 4 706 280 $€_{2015}$ per year and unit for switching to heavy fuel oil with a lower sulphur content.

Figure 7.16 compares the private costs and societal benefits at European level resulting from the installation of primary emission control per power plant unit, showing that costs exceed benefits by about 3.6 million €$_{2015}$ per year under the given conditions. Even before entering the annual data into the CBA decision rule, thereby considering the complete project lifetime, the primary emission control measures can be classified as disproportionate, mainly owing to the high costs of reducing the fuel sulphur content. Applying the CBA decision rule (equation 3.1) results in a net present value of -49 537 143 €$_{2015}$, confirming that the investment is not socially efficient. Whilst it would be interesting to separately analyse the two primary emission reduction measures treated as a bundle here, the available emission scenarios do not allow such a separate analysis.

In order for annualised costs to break even with annual health benefits, the costs of fuel switching would have to decrease to a level of 1 061 250 €$_{2015}$ annually, all other elements remaining equal. This would imply marginal costs of 5 896 €$_{2015}$ per tonne of SO_2 removed and a substantial reduction when compared to the current default value (26 146 €$_{2015}$ per tonne of SO_2 removed). Alternatively, for a break even, the health benefits would need to rise to 19.5 €$_{2015}$ per MWh$_{el}$ at the European scale.

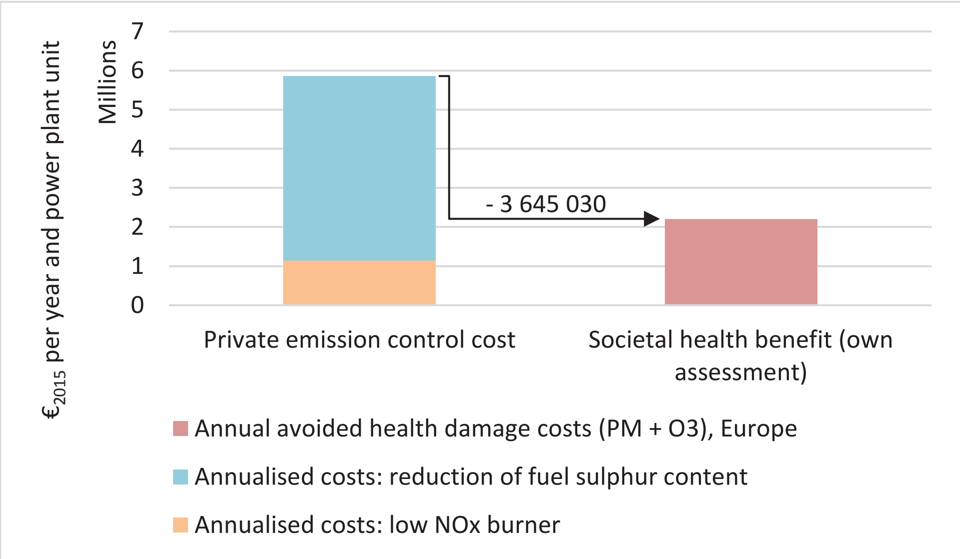


Figure 7.16: Annualised costs of primary emission control measures and related benefits through avoided health damage costs per power plant unit and for the European modelling domain

The benefits of installing a secondary PM emission control measure, i.e. an electrostatic precipitator (ESP), can be derived from the health damage costs displayed in Figure 7.17.

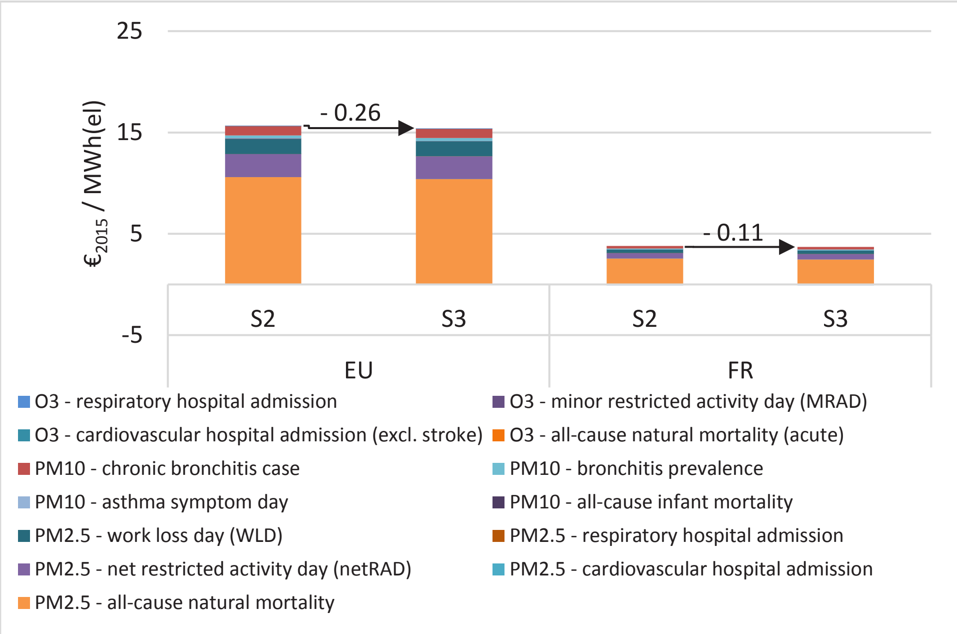


Figure 7.17: Annual health damage costs related to the initial emission level with primary NO_x and SO_2 control measures installed (S2) and after installing an additional secondary emission control measure for PM (S3) at different modelling domains

The reduction in health damage costs amounts to around 0.26 $€_{2015}$ per MWh_{el}. Combining these with 300 000 MWh_{el} generated per power plant unit leads to annual benefits of 78 872 $€_{2015}$. When compared to the annualised cost of the ESP, amounting to 471 664 $€_{2015}$ per unit and year (cf. section 6.2.2.3), costs exceed benefits by 392 792 $€_{2015}$ per unit and year (Figure 7.18). Applying the CBA decision rule (cf. equation 3.1) results in a net present value of -5 338 178 $€_{2015}$, implying that the investment is not socially efficient.

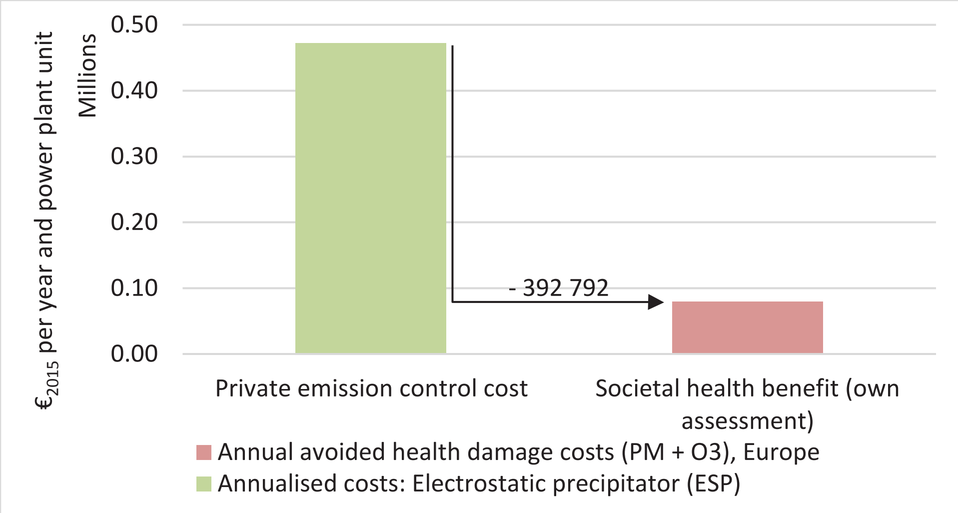


Figure 7.18: Annualised costs of secondary emission control and related benefits through avoided health damage costs per power plant unit and for the European modelling domain

7.3.2 Comparison with simplified approaches for benefit assessment

The health damage cost assessment approach developed in this thesis enables an advanced consideration of the emission source characteristics and its location, however at the expense of a relatively high computational effort for atmospheric modelling (cf. also section 8.4). For this reason, the results obtained above shall be compared to two simplified approaches for benefit assessment:

- First, the benefit assessment is carried out using adapted EcoSenseWeb modelling results (including updated impact functions and monetary valuation in order to enable a better comparability with the standard social CBA results, cf. sections 4.3.1 and 5.2). This yields health damage costs that are site-specific to some extent but fail to account for variability in the operation pattern;

- Second, the benefits assessment is carried out using national average damage cost factors per quantity of pollutant emitted based on guidance by the European Commission (2006a) within the framework of BAT Reference documents (cf. section 2.4.6). These unit cost factors are dated (and could not be updated retrospectively) and account neither for the specific emission place nor for variability in the operation pattern.

Table 7.10 summarises the objectives and related approaches to compare different variants of the benefit assessment within the social CBA of emission control measures.

Table 7.10: Objectives and research approach for the social CBA of primary emission control measures using simplified methods for health benefit assessment

Objectives	1) What is the outcome of the social CBAs on primary and secondary emission control measures when using the EcoSenseWeb model for the health benefit assessment? 2) What is the outcome of the social CBAs on primary and secondary emission control measures when using unit damage cost factors from European Commission (2006a) for the health benefit assessment?
Approach for objective 1)	
What	Assessing health benefits (reduced damage costs related to $PM_{2.5}$, PM_{10}, and O_3) and associated private costs of primary and secondary emission control measures
Where	Europe (EU28 + 11 non EU countries, larger than the Europe domain defined in section 6.1.1.2)
For	Scenarios S1 (initial emission level) as well as S2 and S3 (reduced emission levels), compared to S0 (baseline), cf. Table 6.2
Using	EcoSenseWeb model, adapted as described in section 4.2 in order to match the input parameters of section 6.1, followed by an adjustment for inflation and income growth (cf. section 6.1.4.2); private cost assessment, cf. section 6.2.2
Approach for objective 2)	
What	Assessing health benefits (reduced damage costs related to $PM_{2.5}$, PM_{10}, and O_3) and associated private costs of primary and secondary emission control measures
Where	Europe (EU28 + 11 non EU countries, larger than the Europe domain defined in section 6.1.1.2)
For	Scenarios S1 (initial emission level) as well as S2 and S3 (reduced emission levels), compared to S0 (baseline), cf. Table 6.2
Using	Unit damage costs as described in (Holland et al. 2005c), adjusted regarding inflation and income growth between the year 2000 and 2015 (cf. section 6.1.4.2); private cost assessment, cf. section 6.2.2

For objective 1, dedicated calculations using the model EcoSenseWeb have been carried out. The input data is consistent with the case presented in section 6.1.1. To allow for a better comparison, health impact functions (including age group and risk group fractions) and monetary valuation parameters have been harmonised following the approach described in chapter 4. This means that the remaining differences in resulting health damage costs are due to:

- Dispersion modelling: The atmospheric model used in EcoSenseWeb is a parameterised version of the EMEP model that does not account for variable operation profiles. Moreover, different background meteorology and emission inventory data is used compared to the modelling framework described in section 6.1;

- Exposure modelling: Given that the modelling domain in EcoSenseWeb is larger than the Europe modelling domain used before, more people are considered for the health impact assessment. Moreover, the statistics on population distribution and numbers are older than those used in the modelling framework described in section 6.1.

For objective 2, unit damage cost factors given in European Commission (2006a) include impacts on human health and crops. To allow for a better comparison and using background information by Holland et al. (2005c), only the human health-related damage costs are included here, amounting to:

- 7 200 $€_{2000}$ per tonne of NO_x emitted in France, translating into 10 475 $€_{2015}$ after inflation and income adaption as described in section 6.1.4.2;

- 8 035 $€_{2000}$ per tonne of SO_2 emitted in France, translating into 11 689 $€_{2015}$ after inflation and income adaption as described in section 6.1.4.2;

- 44 000 $€_{2000}$ per tonne of $PM_{2.5}$ emitted in France, translating into 64 011 $€_{2015}$ after inflation and income adaption as described in section 6.1.4.2.

The differences as compared to the approach described in section 6.1 are as follows:

- Dispersion modelling: The results are based on a parameterised version of the EMEP model that does not account for variable operation profiles. This modelling is similar to the one underlying EcoSenseWeb, however it uses an older version of the EMEP model. Background meteorology and emission inventory data differ as well;

- Exposure modelling: Given that the modelling domain is larger than the Europe modelling domain used before, more people are considered for the health impact assessment. Moreover, the statistics on population distribution and numbers are different from those in section 6.1;

- Health impact assessment: As data harmonisation is not possible here, results differ in terms of considered health endpoints and respective impact functions;

- Monetary valuation: No harmonisation of valuation factors was carried out, apart from adjusting the values for inflation and income growth. Particularly for the valuation of mortality risks due to long-term exposure towards PM, a 20% higher valuation factor is used compared to the approach described in section 6.1.

For the benefit assessment the unit damage cost factors (as indicated above) are multiplied by the amount of tonnes of pollutant avoided through the emission control measures (180 tonnes per year of NO_x and SO_2 and 15 tonnes of $PM_{2.5}$[26]).

The above described approaches lead to the results displayed in Figure 7.19 (primary emission control measures) and Figure 7.20 (secondary emission control measure).

In case of installing the primary emission control measures, both alternative approaches lead to an increase in health damages per tonne of pollutant emitted. Therefore, the societal benefit of reducing emissions is higher when compared to the standard modelling approach ('own assessment'). The highest benefit is observed when relying on the unit damage cost factors by European Commission (2006a). Yet, the private costs of emission control remain globally higher than the associated benefits for all of the approaches compared here and the conclusion of the social CBA does not change, i.e. the pollution control measures are not socially efficient.

For the secondary PM emission control measure, using the simplified assessment approaches again leads to substantially increased societal benefits. When relying on the unit damage cost factors by European Commission (2006a), the investment can be classified as socially efficient according to the social CBA. This is mainly due to the high damage costs per tonne of $PM_{2.5}$ emitted (cf. above). Given that a national average unit damage cost factor is used that does not account for the high height of emissions from the oil-fired power plant, it is likely that this approach overestimates the benefits of emission reduction in this case.

[26] It is assumed that the 18 tonnes of PM avoided consist of 84% $PM_{2.5}$ (Dreiseidler et al. 2000).

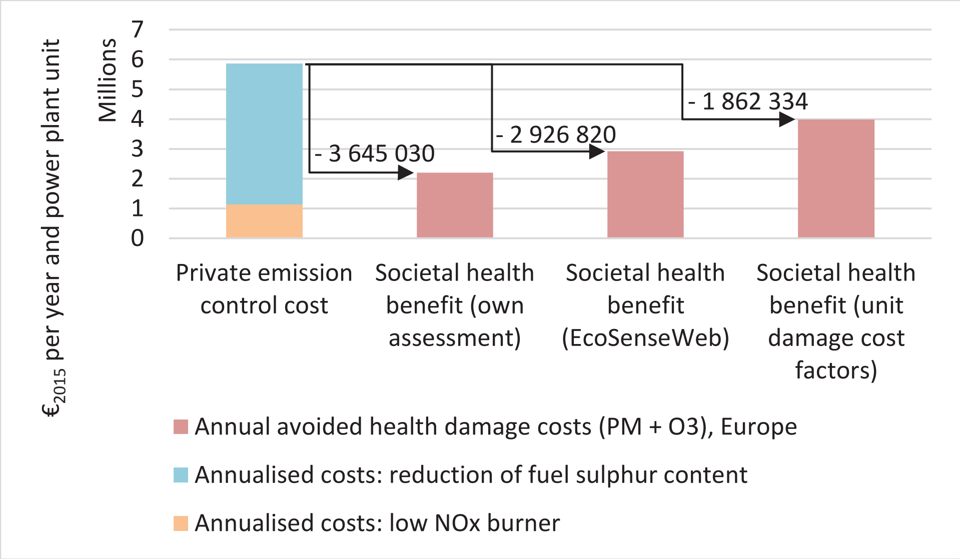


Figure 7.19: Annualised costs of primary emission control measures and related benefits through avoided health damage costs per power plant unit and for the European modelling domain using different approaches for benefit assessment

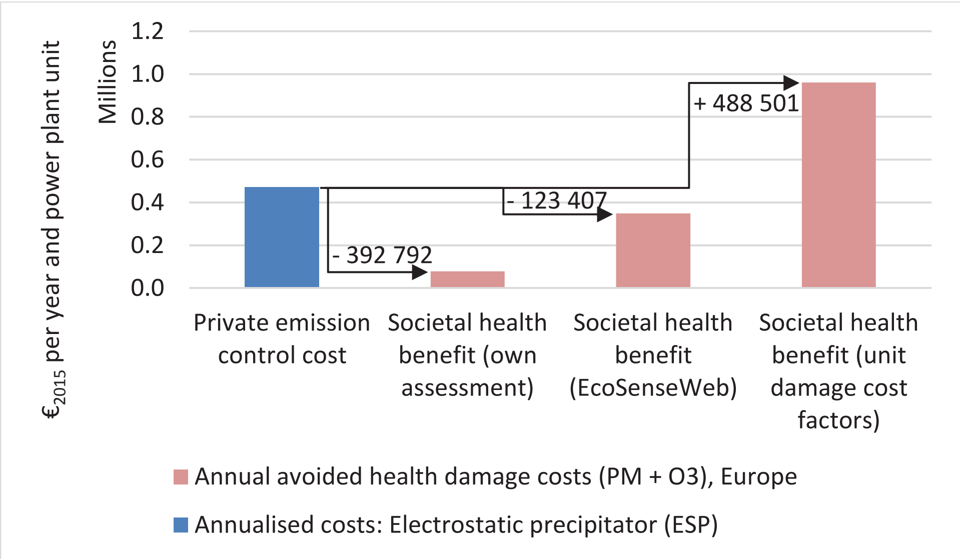


Figure 7.20: Annualised costs of secondary PM emission control and related benefits through avoided health damage costs per power plant unit and for the European modelling domain using different approaches for benefit assessment

7.3.3 Sensitivity analysis

In order to demonstrate the influence of key modelling and parameter choices on the social CBA outcome, a sensitivity analysis is carried out. Given that the methodological choices are interdependent, a step-wise procedure is followed. Based on experience from chapter 4 and sections 7.1 and 7.2, specific methodological choices are analysed. Table 7.11 summarises the objectives and related approaches.

Table 7.11: Objectives and research approach for the sensitivity analysis of input parameters used in the social CBA of primary emission control measures

Objectives	1) What is the outcome of the social CBA on primary emission control measures when varying key modelling choices regarding the benefit assessment? 2) What is the outcome of the social CBA on primary emission control measures when varying key parameters in the emission control cost assessment?
Approach for objective 1)	
What	Assessing health benefits and private costs of primary NO_x and SO_2 emission control measures, when including additional NO_2-related health impacts and when varying the long-term exposure mortality impact assessment (benefit side)
Where	Europe (EU), cf. Figure 6.4
For	Scenarios S1 (initial emission level) and S2 (reduced emission level), compared to S0 (baseline), cf. Table 6.2
Using	Standard modelling framework as described in section 6.1 and private cost assessment described in sections 6.2.2.1 and 6.2.2.2, varying modelling choices for the benefit assessment, described above
Approach for objective 2)	
What	Assessing health benefits and private costs of primary NO_x and SO_2 emission control measures, when varying the equipment lifetime and the weighted average cost of capital (cost side)
Where	Europe (EU), cf. Figure 6.4
For	Scenarios S1 (initial emission level) and S2 (reduced emission level), compared to S0 (baseline), cf. Table 6.2
Using	Standard modelling framework as described in section 6.1 and private cost assessment described in sections 6.2.2.1 and 6.2.2.2, varying input parameters for the cost assessment, described above

As results are similar for the primary and secondary emission control measures, only the former are presented here.

The following adjustments are carried out regarding the benefit assessment:

- inclusion of direct NO_2-related health impacts;
- use of a VSL-based approach for the assessment of long-term exposure mortality impacts (both due to $PM_{2.5}$ and NO_2).

Other important influencing factors have been disregarded for the following reasons:

- Using the plume-in-grid modelling instead of the default modelling (cf. section 7.1.4) was disregarded given that the results are not available at the European domain and that the confidence in the results is not necessarily higher than for the standard modelling (cf. section 8.1.1);
- Using the constant instead of the variable emission scenario (cf. section 7.1.3) was likewise disregarded because such a constant scenario is not realistic for a peak load power plant.

When using the annuity method (cf. section 3.3.1) for the private cost assessment, the annuity is defined by the capital recovery factor that in turn depends on the project lifetime and interest rate (weighted average cost of capital), as displayed in Figure 7.21.

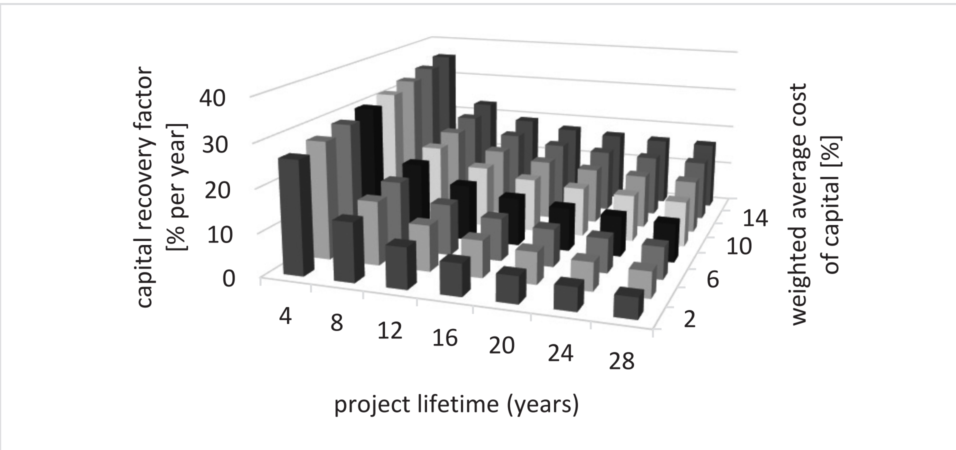


Figure 7.21: The capital recovery factor of the annuity method as a function of project lifetime and weighted average cost of capital for typical ranges of values

Short project lifetimes lead to a substantial increase of the capital recovery factor. As a consequence, annuities increase with shorter project lifetime and with higher weighted average costs of capital. On this basis and by considering further information, the following parameter choices have been made for the sensitivity analysis regarding the private cost assessment:

- Equipment lifetime: The annuities of the low NO_x burner are recalculated using an assumed life time of 10 years (instead of 20 years);
- Weighted average cost of capital: Recommendations given in a recent European reference document (Capros et al. 2016) for 'companies in competitive energy supply markets' are followed, leading to a weighted average cost of capital of 8.5% (instead of 6%).

The cost of fuel switching for SO_2 emission reduction includes only a variable component and is therefore independent from the assumed equipment lifetime and the weighted average cost of capital.

Figure 7.22 shows the result of applying the above listed sensitivity assumptions.

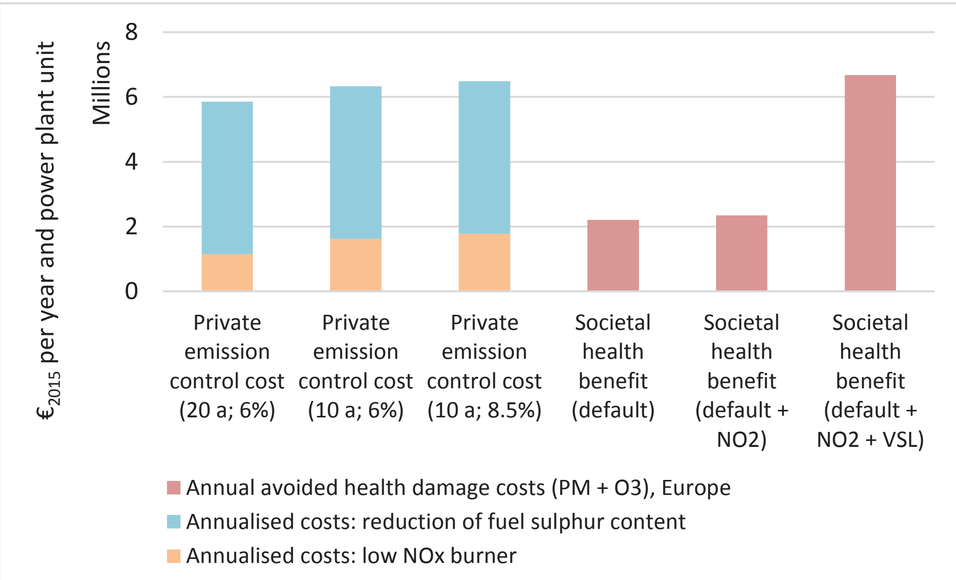


Figure 7.22: Annualised costs of primary emission control measures and related benefits through avoided health damage costs per power plant unit and for the European modelling domain using different assumptions and parameters for the cost and benefit assessment

On the cost side, a decrease of the project lifetime as well as an increase of the weighted average cost of capital both lead to an increase of annualised costs. On the benefit side, considering direct NO2-related health benefits leads only to a slight increase of benefits, whereas switching to a VSL-based approach for the assessment of long-term exposure mortality risk assessment leads to a very substantial increase in health benefits. In the latter case, the CBA would yield a positive net present value for all sensitivity cases analysed for the private cost assessment.

7.4 Summary of chapter 7

The results of several case studies are presented, providing answers to the following overarching objectives: exploring the influence of 1) emission patterns and atmospheric modelling features, and 2) of methodological choices in health impact assessment, as well as 3) demonstrating the application of the methodological framework for social cost-benefit analysis of emission control measures at site level. Using deterministic modelling approaches, the most remarkable observed influence on quantified damage costs concerns the temporal emission pattern (and hence temporal modelling resolution), the use of a plume-in-grid modelling approach, and the approach for long-term exposure mortality impact assessment. Neither the primary nor the secondary emission control measures are socially efficient according to the default assessment following the methodology developed within this thesis. However, using an alternative mortality assessment approach or relying on simplified benefit assessment approaches would, in some cases, alter the outcome of the social cost-benefit analysis.

8 Discussion

The first section of this chapter discusses the case studies of chapter 7 in view of providing further contextual information (cf. section 8.1). Apart from demonstrating the feasibility of social cost-benefit analysis (CBA) at the level of one individual industrial emission source, another merit of these case studies is to provide insights into the influence of methodological choices on health damage costs and, more generally, on the results of social CBAs. As these insights relate to the topic of uncertainty, a large part of this chapter is devoted to uncertainty underlying health damage costs: first, existing evidence on uncertainty is reviewed and put into perspective (cf. section 8.2.1); second, a quantitative (cf. section 8.2.2) and qualitative (cf. section 8.2.3) uncertainty characterisation is presented, largely based on the case studies of this thesis; last but not least, the implications of uncertainty on decision-making in the business and public policy context are discussed (cf. section 8.3). A critical appraisal of the developed methodology beyond uncertainty considerations serves to identify future research opportunities (cf. section 8.4). Based on the experience with transferring the social CBA method from the public to the private context, a set of methodological recommendations for private decision-makers concludes this chapter (cf. section 8.5).

8.1 Discussion of the case study results

Following the structure of chapter 7, the results of the case studies are discussed. The influence of methodological choices on health damage costs is discussed mainly in terms of the underlying mechanisms and put into perspective with related existing work. For the exemplary social CBA, the discussion focuses on the relevance and the reliability of the results.

8.1.1 Influence of atmospheric modelling features

Globally, it is difficult to generalise findings that are related to atmospheric modelling, given the non-linearity and thresholds involved in atmospheric chemistry. Moreover, as discussed below, the modelling results depend on the specific emission source, time, place, and background conditions.

In the specific cases analysed in chapter 7, all results are influenced by the fact that the emission source is located near a large metropolitan area (i.e. the city of Paris), characterised by a high population density, and surrounding areas with a lower population density (Figure 6.6). This spatial variability in population density has a crucial influence on population exposure and resulting health damage costs. It also implies that the sometimes substantial differences due to different modelling features at the local level (Île-de-France domain) would be less pronounced in the case of a less densely populated area near the emission source. Apart from higher population densities, metropolitan areas are typically also characterised by higher background emission levels, particularly due to road traffic. This has an influence on NO_2-related impacts and on the formation mechanisms of secondary pollutants, notably secondary PM or ozone, further discussed below.

Emission intensity

The differences in health damage costs due to different emission intensities were moderate (i.e. ranging from 3% at the Europe domain to 27% at the Île-de-France domain, cf. section 7.1.1). Evidence exists that numerical modelling uncertainty reduces with an increasing emission intensity (Brandt et al. 2012, Tarrasón 2009). As a consequence, the health damage costs per MWh for the constant emission scenario at higher emission intensity can be considered more robust than those at lower emission intensity.

Background emission inventory

Using a different background emission inventory at national level only had a minor influence on the estimated health damage costs (cf. section 7.1.2). Yet, this result strongly depends on the place of emission as well as the substances concerned (those emitted and those already present in the background), especially with regard to the formation of secondary PM.

The robustness of results at local level could be further improved by increasing the spatial resolution of the background emission inventory. In the modelling framework described in section 6.1 and used for the case study, the inventory has a resolution of 50 km, complemented by some additional information derived from land use data. Work by Fountoukis et al. (2013) on the Paris metropolitan region, albeit using a different air quality model than the one in this thesis, has shown that increasing the spatial resolution of the emission inventory leads to larger spatial concentration gradients, thereby improving the precision of exposure estimates.

Temporal modelling resolution

The average modelling approach (constant operation) leads to considerably higher health damage costs than the more highly resolved modelling approach (variable operation) for the case study presented in section 7.1.3. As the annual emission quantities are equal in both scenarios, the underlying causes can be generally attributed to the atmospheric modelling, including the influence of meteorology and background emissions. These lead to a considerably different population exposure for which three reasons are identified.

An analysis of exposure data under the constant emission scenario reveals a seasonal influence regarding PM formation. Notably, under the given conditions and when comparing monthly average exposure data, the same quantity of power plant emissions leads to more pronounced ambient $PM_{2.5}$ concentration increases during the summer months than during the winter months[27]. Combined with the fact that, under the variable emission scenario, the power plant operates mainly during the winter months (Figure 6.2), this helps to partly explain the observed differences between the two operation scenarios.

Second, looking at the wind directions for both scenarios (Figure 7.5) reveals that the constant emission scenario is characterised by prevailing westerly winds, whereas wind directions are more diverse in the variable emission scenario. Given that a large metropolitan area is situated eastwards of the emission source, this partly explains the higher damage costs observed in the constant emission scenario, particularly at the Île-de-France domain.

Another, though minor part of the differences between the constant and variable emission scenario is explained by differences in ozone-related health damage costs. Tropospheric ozone impacts are assumed to occur only above a given concentration threshold. In addition, ozone forms in a non-linear reaction mainly during the summer season. As a result, the operating time of the emission source as well as the meteorological and background emission conditions during the year are relevant influencing factors. In the given case, this leads to lower ozone-related health damages in the variable emission scenario, in which emissions mainly occur during the winter season (Figure 6.2). According to the European Environment Agency[28], the meteorological year 2009, used for the modelling, was characterised by very low summer ozone concentrations compared to other years.

[27] Results available upon request.

[28] http://www.eea.europa.eu/highlights/summer-ozone-record-low-concentrations-in-2009, last accessed: 2017-05-18

Spatial modelling resolution

The case study presented in section 7.1.4 shows that the direction of change in health damage costs with regard to the spatial modelling resolution depends on the analysed zone as well as the resolution itself: increasing the spatial resolution at the France domain (from 35 km x 50 km to 17 km x 25 km) led to a 9% reduction in health damage costs. At the Île-de-France domain, the same increase in resolution slightly increased health damage costs, whereas moving to a high (3 km x 5 km) resolution slightly reduced health damage costs. The latter can be explained by the fact that the high modelling resolution leads to lower concentration levels surrounding the emission source, where large shares of the exposed population are located (cf. Figure 7.8). A similar explanation applies to the France domain, where the averaging of concentrations over a larger area using a lower modelling resolution leads to a higher exposure level in the metropolitan area (results not included graphically in this thesis).

These findings are partly consistent with further work on the same topic. Likhvar et al. (2015) evaluated the influence of spatial modelling resolution on $PM_{2.5}$ and ozone-related health impacts at a European (50 km resolution) and Île-de-France (4 km resolution) domain. However, the analysis differs from the case presented in this thesis in two central points: First, a different air quality model (CHIMERE) was used. Second, Likhvar et al. (2015) assessed the impacts of large-scale emission changes assumed to occur under future European energy scenarios rather than of specific point emission sources. The results (ibid., p. 447) indicate that "*working at the IdF* [Île-de-France] *scale, the results* [for $PM_{2.5}$] *were more than 20% larger than those estimated at the European scale for the same area of interest. Here the discrepancy can be mainly due to the resolution change.*" Moreover, it was found that the effect of modelling resolution on ozone-related impacts was more pronounced than for $PM_{2.5}$ and that ozone-related effects were strongly influenced by the presence of a large metropolitan area.

Thompson et al. (2014) analysed the effects of different emission reduction scenarios on $PM_{2.5}$ and ozone-related health impacts in 9 US regions at 36, 12, and 4 km modelling resolution. Here again, a different air quality model (CAMx) as well as different emission source characteristics (i.e. evaluating emission changes between the year 2005 and 2014) were used. Regarding $PM_{2.5}$, it is found that the influence of spatial modelling resolution on health impacts is different for primary and secondary PM species. While a higher resolution generally resulted in higher impacts due to primary PM, the direction of change for secondary PM depended on the specific region. Overall, $PM_{2.5}$ related impacts were likewise found to be less sensitive towards spatial modelling resolution than ozone-related impacts.

Increasing the spatial resolution through a plume-in-grid modelling approach led to remarkably higher concentration levels when compared to the standard modelling approach (cf. section 7.1.4). Yet, it is not possible to state which of the two modelling approaches performs globally better, as the precision of the respective modelling approaches depends on the specific emission scenario and the modelling parameterisations used (Kim et al. 2014). In order to assess the model performance, modelled concentrations using both approaches would need to be compared to surface monitoring data. This was out of scope of the current thesis.

Other findings concerning the plume-in-grid modelling of this thesis are consistent with observations made by Kim et al. (2014). First, plume-in-grid impacts are less pronounced at coarser modelling resolutions due to a higher dilution of the pollution plume. Second, under a plume-in-grid approach, effects from point sources tend to be lower near the point source and more pronounced further downwind from the source (cf. Figure 7.9).

8.1.2 Influence of methodological choices in health impact assessment

Methodological choices related to health impact assessment influence the magnitude of health damage costs (cf. section 7.2). Similarly to atmospheric modelling features, differences in results due to quantifying direct NO_2-related health impacts and due to changing the ozone concentration threshold depend on the emission source characteristics and the population distribution. By contrast, the differences in results observed due to different mortality impact assessment approaches are independent from source characteristics and population distribution and are therefore more generally valid.

Inclusion of NO_2-related health impacts

Like for $PM_{2.5}$-related endpoints, the increase in mortality risk due to long-term exposure above a concentration threshold of 20 µg of NO_2/m^3 has the highest influence on health damage costs among all NO_2-related health endpoints (Figure 7.10). For this reason the choice of the threshold is of crucial importance. Even though the 20 µg/m³ threshold is officially recommended by the WHO (2013a), it is nonetheless discussed as potentially being too high, given that health effects have also been shown to occur at lower concentration levels (Héroux et al. 2015, Walton et al. 2015). Keeping in mind the quite limited areas for which the threshold was actually exceeded in section 7.2.1 (cf. Figure 7.12), it is clear that decreasing the threshold would lead to a potentially important increase in health damage costs.

Another important influencing factor related to this concentration threshold is the proximity of a large metropolitan area. As such areas are typically characterised by a high volume of road traffic that causes substantial NO_2 emissions, the probability of exceeding a threshold of 20 µg/m³ is much higher, as illustrated by the case study result (cf. Figure 7.11).

Still related to the threshold, the spatial modelling resolution also plays a decisive role. The higher this resolution, the higher the chances to capture local exceedances of thresholds, as concentrations get more diluted when being averaged over a larger area. This effect can be observed for the European modelling domain. The concentration increase estimated over the metropolitan area is lower when compared to the higher resolution modelling at the Île-de-France modelling domain (cf. section 7.2.1). The effects at European level thus tend to be underestimated in the presented case. Indeed, measurement data for 2009[29] shows local exceedances of the 20 µg/m³ concentration threshold at many places all over Europe, while these exceedances are not identified by the atmospheric modelling at the European domain. As a consequence, in order to obtain more precise results at European level, a higher modelling resolution would be needed for countries other than France. This is however not deemed necessary, given that the exemplary emission source is located quite centrally within France. Therefore, the majority of the total effects is expected to be captured by the higher modelling resolutions at the Île-de-France and France domains.

In addition to the questions surrounding the appropriate effect threshold, there is a risk of double counting related to long-term exposure mortality impacts, if effects due to $PM_{2.5}$ exposure and additionally due to NO_2 exposure are estimated. According to WHO (2013a), the overlap is said to be in the range from 0 to 33%. For this reason and taking a rather conservative approach, only 67% of the direct NO_2-related mortality health damage costs were considered in section 7.2.1 and in section 7.3.3.

Using an alternative approach for long-term exposure mortality impact assessment

The way in which long-term exposure mortality risks are assessed is the single most influential factor within health damage costs assessment. It includes the choice of the approach as such (i.e. VOLY versus VSL) and of the valuation parameter to be used (i.e. how to value a VOLY or VSL).

[29] See http://www.eea.europa.eu/themes/air/interactive/no2, last accessed: 2017-05-18

As shown in section 7.2.2, using a VSL-based approach instead of a VOLY-based approach increases related health damage costs by at least a factor of 3.1. This outcome also depends on the health baseline rates and age group fractions used, explaining the differences between the French and European modelling domain. At the same time, the central VSL estimate of 1.9 million €$_{2015}$ (net of income adaption) used, can be characterised as rather conservative when compared to alternative values. For instance, the OECD (2012) proposes a value of 3.6 million US$$_{2005}$ for the EU context; the U.S. EPA proposes a central VSL estimate of 7.9 million US$$_{2008}$ for policy analysis (U.S. EPA 2010). This shows an important discrepancy between what is currently recommended at both sides of the Atlantic with potentially important policy implications.

The impact of using different VOLY estimates on health damage costs caused by an exemplary power plant has been analysed in section 4.4.1. An increase of this parameter by about 30% leads to an increase in overall health damage costs of about 20%. Although this difference does not seem substantial, it may be sufficient to shift the binary efficiency decision criterion based on a social CBA in either direction.

Lowering the concentration threshold for ozone-related health impacts

Ozone-related health impacts are strongly influenced by the presence of large agglomerations (Likhvar et al. 2015, Markakis et al. 2014, Thompson et al. 2014). Owing to the non-linearity in ozone formation, ozone-related health impacts as estimated in section 7.2.3 additionally depend on the background emission inventory, meteorology, the spatial and temporal modelling resolution, and, as illustrated, the chosen threshold level. The strong sensitivity of results with regard to changes in concentration threshold and modelling domain also imply a generally higher uncertainty of ozone-related health damage costs. Given however that ozone-related health damage costs are minor when compared to those related to PM or NO_2, the effect of these uncertainties on decision-making is limited.

8.1.3 Social CBA of emission control measures

Regarding the practical relevance of the social CBA results (cf. section 7.3) for private decision-making, it should be noted that the emission levels obtained after retrofitting emission control measures (cf. Table 6.2) comply with requirements of the Industrial Emission Directive (IED, cf. section 2.4.6), applicable in the year 2016. For power plants operating less than 1500 full load hours per year, however, derogations are foreseen within Annex V of the IED (European Parliament and Council of the European Union 2010), resulting in less stringent emission limit levels than those assumed in the social CBA. A further tightening of emission limit values in the future will result from the recently concluded revision of the BREF on large combustion plants (cf. section 2.4.6). Due to uncertainty regarding the national implementations and potential flexibilities, the social CBA carried out in section 7.3 is to be seen as a proof of concept rather than being concerned with decision-making under realistic conditions.

While both the primary and secondary emission control measures assessed were qualified as disproportionate using the highly resolved modelling approach implemented in this thesis, the outcome for the secondary control measure changed when using a simplified approach for the benefit assessment (cf. section 7.3.2). As already mentioned, the unit cost factors for $PM_{2.5}$ in the simplified approach do neither account for the specific location and height of the emission source nor for variability in the operation pattern. Moreover, given that they were derived from an older impact pathway implementation with dated risk and valuation parameters, they cannot be considered a robust choice, even though they stem from an official European Commission reference document.

The sensitivity analysis revealed that changing parameters for the private cost assessment had a relatively limited influence, whereas changes in the benefit assessment methodology were more influential. While the inclusion of NO_2-related health impacts had a fairly limited impact on the overall benefits, switching from a VOLY- to a VSL-based mortality impact assessment altered the final outcome of the social CBA.

For a critical appraisal of the social CBA methodology, refer to section 8.4.3.

8.2 Uncertainties in health damage cost assessment: quantitative and qualitative evidence

This section serves to present existing evidence regarding the uncertainties underlying health damage cost assessment in general as well as quantitative and qualitative insights generated through the case studies within this thesis.

8.2.1 Existing evidence on uncertainties underlying health damage cost assessment

As introduced in section 3.4.3, the assessment of health damage costs based on the impact pathway approach is subject to various types of uncertainty. In the European research context, particularly related to the ExternE project series, a few comprehensive uncertainty assessments exist:

- Rabl and Spadaro (1999), Spadaro and Rabl (2008): The central statistical theorem states that, given the multiplicative chain underpinning the impact pathway approach (cf. equation 3.3), the resulting damage costs are lognormally distributed. Under this condition, the 68% confidence interval around the geometric mean (μ_g) can be conveniently expressed using the geometric standard deviation (σ_g), i.e. spanning a range of [μ_g * 1/σ_g ; μ_g * σ_g]. For instance, the following estimates for σ_g are given for mortality-related health damage costs due to directly emitted PM (Spadaro and Rabl 2008):

    - Exposure modelling (including dispersion, chemical transformation and background emissions): σ_g = 1.5 for non-reactive primary pollutants (for secondary PM in the form of sulfate or nitrate σ_g amounts to respectively 1.76 and 1.9);

    - Health impact assessment (including relative risk, PM component toxicity, quantification of YOLL): σ_g = 1.88;

    - Monetary valuation of YOLL: σ_g = 2 for valuation based on WTP from surveys (for market-based valuations σ_g is assumed to be in a range of 1.1 to 1.3).

  These assumptions lead to a σ_g of 2.78 for mortality-related health damage costs due to direct PM emissions. For emissions of SO_2 and NO_x, as precursors for secondary PM-related mortality impacts, σ_g amounts to respectively 3.42 and 3.55. Based on this and some further results, the authors (ibid.) conclude that inhalation-related health damage costs due to emissions of classical air pollutants can be characterised by a σ_g of approximately 3.

  It should be noted that the algebraic models underlying these uncertainty estimates are relatively simple and not comparable to the atmospheric models used in this thesis and that some of the assumptions taken, e.g. on differing PM component toxicity for secondary PM, are not consistent with latest WHO recommendations (cf. section 3.4.1). Yet, these results are useful in that they provide order of magnitude values of uncertainty ranges for central parameters during different assessment stages and for different substances.

- Holland et al. (2005b): This relatively broad uncertainty assessment was carried out in the frame of the social CBA underpinning the Clean Air For Europe (CAFE) programme, i.e. an evaluation of air quality legislation at the European level. It includes a probabilistic uncertainty assessment using Monte Carlo analysis, sensitivity analysis and a qualitative discussion of possible biases. For instance, based on probabilistic Monte Carlo analysis, the 95% confidence interval of the so-called aggregate damage function[30] for PM-related health impacts is found to range from approximately a factor of 2.5 below to a factor of 1.7 above the mean value.

  A critical step during Monte Carlo analysis is the definition of input data (parameter estimates and corresponding probability distribution functions), given that the results are directly impacted by these assumptions. In the present case (ibid.), these assumptions were partly taken on subjective grounds and should therefore be interpreted with caution. The sensitivity analyses address similar elements as this thesis, notably the approach for mortality impact assessment or different effect thresholds for ozone, however at a more aggregated level. Further on, the influence of the choice of the meteorological year is addressed: results indicate that the variation in exposure due to differences in meteorology at the national level are up to 50%, but are reduced when taking a pan-European perspective.

  Although building upon the impact pathway approach, the parameter choices and further assumptions used for these uncertainty analyses are outdated compared to the current thesis.

- Holland (2014b): Closely related to the HRAPIE project (WHO 2013a) and the recent European Clean Air Policy Package (cf. section 2.4.5), this report includes a rather brief and incomplete quantitative and qualitative uncertainty assessment of health-related damage costs. The 95% confidence intervals around the central HRAPIE health risk parameters as well as subjective indications of confidence intervals and probability distributions for population at risk and incidence rates are presented. Moreover, based on the assumed confidence intervals of alternative

[30] The aggregate damage function represents the health damage costs per person and ambient pollutant concentration increment, thus excluding exposure modelling-related uncertainties.

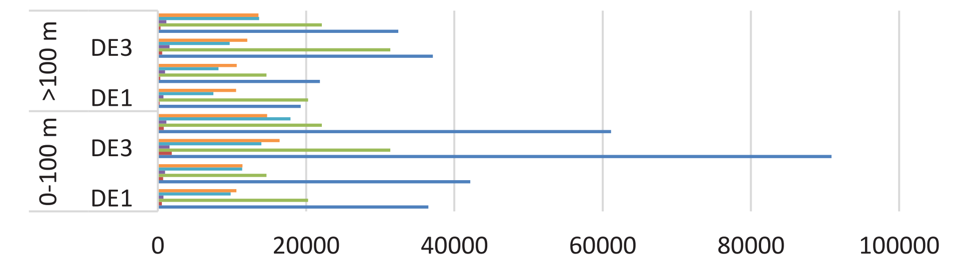
monetary valuation factors for mortality impacts, an uncertainty framework for social CBA is presented that aims at estimating the probability of benefits exceeding costs. On a qualitative level, a bias assessment for unquantified uncertainties is also included.

Regarding the central parameters used for health impact assessment and monetary valuation, most choices correspond to the ones used in this thesis. By contrast, Holland (2014b) uses a higher default monetary valuation for mortality-related impacts compared to this thesis (cf. also section 4.3.5) and disregards atmospheric modelling-related aspects.

8.2.2 Quantitative uncertainty characterisation based on the results of this thesis

Health damage costs caused by atmospheric emissions from point emission sources are influenced by numerous influencing factors, as demonstrated in this thesis. The variation in health damage costs brought about by changing specific influencing factors can be compared on a quantitative basis and delivers new insights for decision-making. In terms of atmospheric modelling and particularly when assessing one individual emission source, the analyses cover more aspects than the existing uncertainty analyses discussed above.

In order to capture the influence of different emission source characteristics more broadly, Figure 8.1 reproduces recent work carried out in the frame of updating the German methodological convention on environmental costs (van der Kamp et al. forthcoming). The associated health damage costs are based on the EcoSenseWeb model and have been updated to reflect recent recommendations in the same way as described in section 4.2 (using the WHO HRAPIE recommendations for impact assessment and monetary valuation according to the "Year 2013" implementation, however updating the monetary base year to 2015).



	0-100 m				>100 m			
	DE1	DE2	DE3	DE4	DE1	DE2	DE3	DE4
SO2	10578	11402	16436	14721	10545	10643	12082	13584
NOx	9813	11364	13983	17887	7487	8187	9686	13646
NMVOC	746	984	1564	1128	746	984	1564	1128
NH3	20282	14638	31375	22142	20282	14638	31375	22142
PPMcoarse	555	724	1898	812	229	317	590	378
PPM2.5	36501	42160	90873	61146	19285	21846	37091	32422

Health damage costs in $€_{2015}$ per tonne emitted

Figure 8.1: Health damage costs per tonne of substance emitted in 4 German sub regions (DE1 – DE4) and per height of release; adapted from van der Kamp et al. (forthcoming)

In terms of their generalisability, the influencing factors on health damage costs, presented below (Table 8.1), can be distinguished into two groups:

- Input data and methodological features related to atmospheric modelling: Input data includes emission source characteristics, such as location, height of release, and emission pattern that, in conjunction with background emission inventory and meteorological data, determine the variations in ambient pollutant concentrations. Due to the non-linear processes underlying atmospheric chemistry, it is generally not possible to derive general conclusions from related analyses, but rather to present the range of possible results per setting (cf. Figure 8.1). Choices regarding spatial and temporal resolution have an influence on the precision of the results, but the direction and size of their influence depends likewise on the specific case and cannot be generalised (cf., for instance, section 7.1.4).

- Elements that are related to the remaining steps of the impact pathway approach, i.e. exposure modelling, health impact assessment, and monetary valuation. Given the linear assessment chain used for the implementation in this thesis (equation 3.3), the influence of these elements on the results can be generalised.

The bulk of analyses carried out within this thesis concerns the first group, i.e. the atmospheric modelling inputs and features.

To characterise the sensitivity of health damage costs with regard to the variation per influencing factor, relative differences are calculated, i.e. dividing a (higher) reference value by a (lower) baseline value. Given however that these sensitivities refer to results that have been calculated using different models and methodologies, they cannot be directly compared. Rather, the intention is to provide order of magnitude values for the influence of single elements (input data and methodological choices) on health damage costs. For reasons of completeness, Table 8.1 also contains influencing factors for which no specific analysis has been carried out in the frame of this thesis.

Next to presenting the outcome of the sensitivity analyses, Table 8.1 also indicates which types of uncertainty are underlying the respective influencing factor, following the classification of section 3.4.3.

Table 8.1: Overview on influencing factors, underlying types of uncertainty and sensitivity of health damage costs with regard to changing the influencing factors; based on case studies within this thesis or related work

Influencing factor	Variability	Parameter uncertainty	Model uncertainty	Decision-rule uncertainty	Sensitivity of health damage costs with regard to changes in the influencing factor	Reference
Atmospheric modelling: emission source characterisation and modelling features						
Emission source: geographic location	x	x	x		**Increase of a factor of 1.8** when locating a power plant (high height of release) at a densely populated area compared to a remote location (low population density); European assessment domain	section 5.4.2
	x	x	x		Differences of **up to a factor of 2.49** between different regions in Germany for direct $PM_{2.5}$ emissions at a low height of release (regional modelling); European assessment domain	Figure 8.1

Influencing factor	Variability	Parameter uncertainty	Model uncertainty	Decision-rule uncertainty	Sensitivity of health damage costs with regard to changes in the influencing factor	Reference
Emission source: height of release	x	x	x		**Increase of up to a factor of 2.45** when moving from a high to a low height of release for direct $PM_{2.5}$ emissions in a given German region (regional modelling); European assessment domain	Figure 8.1
Emission source: substances emitted	x	x	x		Considerable variations per tonne of primary substance emitted. E.g. damage costs per tonne of primary $PM_{2.5}$ **up to a factor of 58 higher** than those per tonne of NMVOC at a low height of release in a given German region; European assessment domain	Figure 8.1
Emission source: emission intensity	x	x	x		Decrease of **between 3%** (Europe modelling domain) **and 27%** (Île-de-France modelling domain) when moving from low to high emission intensity	section 7.1.1
Background emission inventory	x	x	x	x	**Increase of 3%** at the France modelling domain between the year 2009 and a hypothetical inventory of the year 2020	section 7.1.2
Use of plume-in-grid model	x	x	x	x	Increase of **between a factor of 1.75** (France modelling domain) **and 2.51** (Île-de-France modelling domain) when using the plume-in-grid modelling	section 7.1.4
Spatial modelling resolution	x	x	x		**Decrease of 9%** when moving to a higher resolution (France modelling domain); direction of change depending on the chosen resolution at Île-de-France modelling domain	section 7.1.4

Influencing factor	Variability	Parameter uncertainty	Model uncertainty	Decision-rule uncertainty	Sensitivity of health damage costs with regard to changes in the influencing factor	Reference
Temporal modelling resolution/ emission pattern	x	x	x		Increase of **between a factor of 1.69** (Europe modelling domain) **and 4.11** (Île-de-France modelling domain) when moving from an average (constant emission pattern) to a specific modelling (variable emission pattern)	section 7.1.3
Exposure assessment						
Population density	x	x	x		See above (Emission source: geographic location)	section 5.4.2
Health impact assessment						
Concentration-response function		x	x		Not analysed separately in chapter 7; highest influence for endpoints with important contribution to overall damage costs (i.e. mortality impacts due to long-term $PM_{2.5}$ exposure, cf. below)	section 4.3.4
Health baseline rates and age group fractions	x	x			Not analysed separately here; highest influence for endpoints with important contribution to overall damage costs (i.e. mortality impacts due to long-term $PM_{2.5}$ exposure, cf. next item)	n. a.
Approach for mortality impact assessment		x	x	x	Difference of **between a factor of 3.09** (France modelling domain) and **3.69** (Europe modelling domain) when switching from an approach based on years of life lost to an approach based on cases of premature deaths	section 7.2.2

Influencing factor	Variability	Parameter uncertainty	Model uncertainty	Decision-rule uncertainty	Sensitivity of health damage costs with regard to changes in the influencing factor	Reference
Differential PM component toxicity		x	x	x	Increase of **between a factor of 1.47** (remote location) **and 2.02** (densely populated area) when using equal toxicity instead of differential toxicity; European assessment domain; the WHO (2013a) recommends equal PM component toxicity	section 5.4.4.2
Choice of health endpoints		x	x	x	Highest influence for endpoints with important weight in overall damage costs, i.e. all-cause natural mortality due to long-term $PM_{2.5}$ exposure (contributing **around 68%** to the overall quantified damages when excluding direct NO_2-related damages, Europe modelling domain)	section 7.3
		x	x	x	NO_2-related long-term exposure mortality impacts contribute to **between 3%** (European modelling domain) **and 39%** (Île-de-France modelling domain) of total damage costs; strongly dependent on the effect threshold and treatment of double counting with PM effects	section 7.2.1
Monetary valuation						
Use of different base years: inflation and income adjustment		x	x	x	20% increase for an inflation adaptation from the year 2005 to 2015 within the Euro zone; case-specific otherwise. Influence of income adjustment depending on the geographic zone and elasticity of willingness-to-pay with regard to personal income; 5.2% increase for an adaptation from the year 2005 to 2015 in the EU28	section 6.1.4.2

Influencing factor	Variability	Parameter uncertainty	Model uncertainty	Decision-rule uncertainty	Sensitivity of health damage costs with regard to changes in the influencing factor	Reference
Discounting		x	x	x	Influence depending on assumptions concerning the discount rate; no influence on the outcome of the social CBA in section 7.3, given that annualised costs and benefits are constant over the project lifetime and that an equal social discount rate was applied	section 6.3
Valuation per health endpoint	x	x	x	x	Highest influence for endpoints with important contribution to overall damage costs, i.e. all-cause natural mortality due to long-term $PM_{2.5}$ exposure; Different assumptions on the VOLY have led to a 20% difference in health damage costs	section 4.4.1
Valuation components				x	If only tangible (marketable) components were included in the valuation, the benefits of emission control measures would be substantially lower, owing mainly to the fact that mortality-related impacts would be excluded, cf. Table 6.8	section 6.1.4

As a conclusion, the most notable sensitivities according to the methodological choices analysed here are observed concerning:

- Emission source characteristics: Location, height of release as well as the substances concerned are shown to have a high influence on health damage costs. The variation in health damage costs due to differences in these characteristics can be used as a proxy for the error induced when working with generic or average damage cost factors (cf. the simplified benefit assessment approach in section 7.3.2) instead of more specific modelling approaches;

- Temporal modelling resolution and the use of a plume-in-grid approach (related to spatial resolution at local level) are both shown to have a considerable influence on health damage costs. While a higher temporal modelling resolution generally improves the precision of the results, the added value of the plume-in-grid modelling in terms of robustness remains unclear in the absence of a validation with ambient monitoring data;

- PM component toxicity: Although an important influencing factor in theory, the practical relevance is limited as current recommendations are to treat all particle components as equal in terms of toxicity (WHO 2013a). However, due to ongoing discussions on this point (cf. section 8.4.1), additional sensitivity analyses may be envisaged. This requires to differentiate between primary and secondary particle fractions in the results of the atmospheric modelling;

- The choice of the mortality impact assessment approach is crucial given its high influence on the total health damage costs. In the absence of a clear recommendation concerning the preferred approach, it appears reasonable to carry out a sensitivity analysis using alternative assumptions when conducting social CBA.

8.2.3 Qualitative assessment of uncertainties underlying the social CBA on emission control measures

A qualitative bias review of unquantified elements with an impact on the social CBA of emission control measures (cf. section 7.3) is carried out in the following (Table 8.2), including information on:

- Element: what is the unquantified element?

- Impact on social CBA: describing whether the omission is likely to increase or decrease the societal benefit or private cost (ceteris paribus, i.e. all other elements remaining equal);

- Mechanism: description of how the element influences the outcome of the social CBA.

Note that the information presented below is a first step towards a critical appraisal of the methodology developed within this thesis (cf. section 8.4).

Table 8.2: Review of unquantified elements with an impact on the social CBA of emission control measures

Element	Impact on social CBA	Mechanism
Emission source characterisation		
Auto electricity consumption of emission control equipment	Decrease in societal benefits	Relevant mainly for secondary pollution control measures (European Commission 2006b); power plant efficiency decreases, which results in a slight increase in atmospheric emissions and related damage costs per unit of electricity produced.
Power plant efficiency losses due to part-load operation and cycling	Direction unclear	Although efficiency losses lead to higher emission levels per unit of electricity produced for some pollutants (CO_2, SO_2), non-linear relationships exist in case of NO_x, also depending on the power plant type (Lew et al. 2013); the overall effect on health damage costs is thus unclear.
Emissions of trace pollutants	Decrease in societal benefits	Depends on the extent to which emissions of trace pollutants are influenced by the installation of emission control equipment. Damage costs due to trace pollutants are highly site-dependent and their quantification is rather complex (Bachmann 2006).
Emissions of greenhouse gases	Decrease in societal benefits	Depends on the extent to which greenhouse gas emissions are influenced by the installation of emission control equipment. Small increase of damage costs per unit of electricity generated in case of auto electricity consumption or other efficiency losses (cf. above).
Releases into other media, e.g. water or soil	Decrease in societal benefits; not relevant here	Secondary emission control measures may lead to additional discharges, e.g. into water, causing environmental or health impacts. Quantification is highly site-specific and complex.
Solid waste discharge	Direction unclear	Solid waste resulting from certain emission control equipment needs to be discharged and, depending on the method of discharge and potential valorisation, may cause positive or negative effects.
Atmospheric modelling and exposure assessment		
Larger geographical zone	Increase in societal benefits	Extending the assessment to further countries would lead to additional health benefits, however expected to be small in the present case.

Element	Impact on social CBA	Mechanism
Dynamic modelling and exposure assessment	Direction unclear	During the social CBA, modelling results based on one exemplary year are projected statically into the future. Accounting for (expected) changes in meteorology, background emissions, and population characteristics and distribution over time would add to the precision of the results.
Impact assessment		
Effects on crops, building materials and ecosystems	Potentially important increase in societal benefits	Effects limited according to existing quantification methods (cf. section 5.4.2) but potentially important, notably when comprehensively accounting for ecosystem (services).
Additional health endpoints	Increase in societal benefits	As discussed by WHO (2013b), recent evidence exists to quantify further health effects, thereby increasing health impacts of air pollution.
Country-specific health risk parameters and baseline rates	Direction unclear	Effect on health damage costs depending on population characteristics and national health statistics (cf. also section 8.4.1).
Life cycle assessment of emission control equipment	Decrease in societal benefits	Depending on the place and characteristics of for instance production processes, additional environmental and health damage costs occur, that are however out of scope of the current thesis.
Monetary valuation		
Further valuation components, non-use values in particular	Increase in societal benefits	The more components of an environmental good or service are assigned a monetary value, the higher the societal benefit related to protecting this good or service.
Private cost assessment		
More sophisticated private cost assessment	Direction unclear	The costs of emission control measures are specific for each setting and are best assessed by the operators themselves.

Element	Impact on social CBA	Mechanism
Decreasing security of supply due to power plant shut downs	Potentially important increase in societal costs	If security of supply, as a positive externality of flexible power plants, is not sufficiently accounted for by market mechanisms, there is a risk that these plants are shut down due to unprofitability. This can be exacerbated, for instance, by environmental regulations. Related to this, the IED foresees derogations for peak load power plants.
Distortions in markets for emission control measures	Direction unclear	If costs of emission control measures are distorted, e.g. by market failures such as monopolies or state interventions, the outcome of a social CBA may not reflect societal efficiency any more.

8.3 Implications of uncertainty on private and public decision-making

Uncertainties are an obstacle to robust decision-making. Some general remarks regarding uncertainty and decision-making can be made:

- While the uncertainty around single elements may be very high, its impact on decision-making depends on the global influence of the respective elements (cf. section 8.2.2);

- The impact of modelling and parameter uncertainties depends on the way in which results are processed. In case that one single scenario is used to inform decision-making, the possible errors induced by key uncertainty factors are potentially very large. The error is reduced, however, when conducting comparisons between different scenarios (e.g. power plant operation scenarios) and when analysing relative differences, such as during the social CBA of emission control measures (cf. section 7.3);

- The relative weight of different types of uncertainties differs as a function of the analysed case and the methods used, as shown in the following examples:

- For instance, Fountoukis et al. (2013) found that uncertainty underlying input data to atmospheric modelling, such as background emissions or meteorology, lead to larger discrepancies between measured and modelled concentrations than differences in the spatial modelling resolution;
- Thompson et al. (2014) found that the uncertainty represented by the confidence interval around the concentration-response function for assessing PM-related mortality is in a similar order of magnitude as using different modelling resolutions;
- Mansfield et al. (2009) found that decision-rule uncertainty (e.g. selecting endpoints and corresponding risk functions) is more influential than underlying parameter uncertainty (i.e. the confidence intervals around the selected risk functions).

Beyond these general remarks, the different ways in which uncertainty impacts decision-making in the private and public sector shall be discussed.

8.3.1 Implications of uncertainty for private decision-making

Given that private decision-making differs from public decision-making in certain key aspects (e.g. no public consultations, less competing interests from diverse stakeholder groups, etc.), certain safeguard mechanisms that help to reduce the potential consequences of uncertainty on decision-making are lacking. Especially in case of high financial or political stakes, it is therefore crucial to provide the private decision-maker with robust guidance and methodologies.

In order to produce robust outcomes, a key choice is to determine the general approach for health damage cost assessment. As shown in section 7.3.2, using a simplified approach based on a dated methodology induces the risk of obtaining a different social CBA outcome than the more robust specific assessment approach. Given the lack of methodological guidance for the private sector, this thesis discusses how to choose an appropriate assessment approach under different conditions (cf. section 8.5).

While reducing considerable uncertainties is imperative for robust private decision-making, it needs to be weighed against increasing resources and expertise required for conducting specific modelling approaches. As long as official methodological guidance is lacking, the tolerable level of uncertainty thus depends on the decision context and the financial stakes. Regarding the latter and keeping in mind the substantial costs related to power plant retrofits, it can reasonably be expected that conducting a specific modelling exercise in order to obtain a regulatory derogation will in most cases be justifiable.

Regarding the current methodological state-of-the-art, the approach and valuation parameter for assessing long-term exposure-related mortality crucially influence the quantified health damage costs. In this thesis, a rather conservative approach for mortality impact valuation was used by default, i.e. using a valuation parameter that is around 20 % lower than the central parameter used in recently concluded European air quality legislation (cf. section 6.1.4.1). Such a rather cautious approach appears justified from the perspective of the private decision-maker, given the uncertainties related to valuing nontangible goods through surveys. Similarly, a rather cautious approach to quantifying NO_2 exposure-related mortality risks appears reasonable (i.e. using a concentration threshold of 20 µg of NO_2/m^3 and reducing the impact by a third in order to account for possible overlap with PM, cf. section 7.2.1).

8.3.2 Implications of uncertainty for policy decision-making

While uncertainties underlying damage costs are considerable (cf. section 8.2.1), Rabl et al. (2004) state that the risk of cost penalties due to uncertainties underlying damage costs is overall rather small in policy decision-making. Binary (e.g. as in social CBA) and continuous (e.g. when defining environmental quality standards) decision situations are distinguished.

Quite intuitively, the impact of uncertainties on the result of a social CBA (binary decision) is small in cases where the difference between costs and benefits is initially very large. For continuous choices, the effect of uncertainty is likewise said to be rather small "[…] *because near an optimum, the total social cost varies only slowly as individual cost components are varied.* […]" (ibid., page 400). However, these findings cannot be transferred to private decision-making where the results are usually much more sensitive towards single choices, as shown in this thesis (cf. section 8.2.2).

The joint consideration of several individual emission sources or even sectors plays a crucial role in typical policy decision-making. Given the many substances emitted at many different places, the effects of influencing factors are more likely to be evened-out at the aggregated scale than at the scale of a single installation. Moreover, the uncertainties underlying atmospheric modelling are lower as the emission signals are typically much higher (cf. section 8.1). Finally, model uncertainty is typically less critical in policy assessments concerned with comparing (inherently uncertain) future emission scenarios.

Last but not least, while social CBA supports policy decision-making, it is typically used to prepare political negotiations. Given the diverse stakeholders involved in the decision process, these negotiations typically lead to compromises that are less ambitious than the originally recommended objectives based on social CBA[31].

8.4 Critical appraisal of the developed methodology framework and associated research opportunities

First elements for a critical appraisal of the social CBA methodology developed in this thesis have already been presented in section 8.2.3 (i.e. the qualitative bias review of unquantified impacts). The current section aims at providing complementary information regarding these elements and to highlight further methodological constraints as well as associated research opportunities. It distinguishes between the health damage cost assessment, private cost assessment and social CBA.

8.4.1 Health damage cost assessment

By definition, any kind of modelling involves a simplified representation of reality and is thus subject to limitations. This is particularly true when valuing intangible goods such as human health and when modelling complex processes related to atmospheric chemistry, as already discussed in section 8.2. Following the steps of the impact pathway approach, selected methodological shortcomings are discussed below.

Emission source characterisation: consideration of cycling and part-load operation on emission levels

The variable emission scenarios underlying the social CBA in chapter 7 are based on the assumption that emission levels are proportional to the electric load. As a result, efficiency losses due to part-load operation and associated changes in emission levels are ignored. Although this limitation is not deemed too relevant when comparing equally affected scenarios as in this thesis, it is nonetheless interesting to briefly explore the associated consequences.

[31] An example is the recently revised NEC Directive (European Parliament and Council of the European Union 2016), whose final ambition level is lower than the "optimally" defined ambition level based on social CBA (European Commission 2013).

The US Western Wind and Solar Integration Study (Lew et al. 2013) studied the influence of cycling and part-load operation (induced by intermittent renewables) on emission levels of fossil fuel power plants. As a consequence of cycling (load shifts), CO_2 and SO_2 emissions per unit of electricity are found to increase. For NO_x emissions, the case is less obvious. While ramping (i.e. changing the load level) generally leads to higher NO_x emissions, stable part-load operation may avoid NO_x formation when compared to full load operation due to lower levels of (thermal) NO_x formation. However, this effect also depends on the type of power plant technology (Lew et al. 2013).

Whilst being less relevant for social CBA of emission control measures at site-level, the influence of this limitation in social CBA at an aggregate level, e.g. for future policy assessment, could be more substantial. This is due to the increasing flexibility requirements of fossil fuel power plants expected in future energy scenarios (cf. section 2.3).

Atmospheric modelling

Atmospheric modelling based on an Eulerian chemistry transport model as used in this thesis (cf. section 6.1.1) is highly complex and therefore subject to various constraints. Constraints with particular relevance for this thesis are listed below:

- Spatial resolution of background emission inventory: While a relatively high spatial atmospheric modelling resolution at Île-de-France (3 km x 5 km) and French (17 km x 25 km) level was used in the case studies of chapter 7, the resolution of the background emission inventory can be qualified as rather coarse (50 km x 50 km). Even though some adjustments based on land use data were carried out in addition (Legorgeu 2016), the spatial resolution of the emission inventory could be further increased. This would lead to a higher consistency with the grid cell resolution of the atmospheric modelling and, as a consequence, would increase the precision in results (cf. section 8.1.1).

- Plume-in-grid modelling: As shown in this thesis, the influence of using a plume-in-grid approach on health damage costs can be substantial (cf. section 7.1.4). Though consuming a lot of computational resources, such an approach could be justified by the gain in precision in modelled concentrations around the emission source. Yet, the general performance of a plume-in-grid modelling compared to a modelling without plume-in-grid treatment is not automatically higher and depends on various factors, as discussed by Kim et al. (2014). Therefore, further research should aim at increasing the robustness of the plume-in-grid modelling approach and clarify under which conditions such an approach could represent an added value.

- Secondary PM formation: Comparisons of modelled ambient pollutant concentrations based on the Polair3D model (cf. section 6.1.1) with ambient monitoring data have found the largest deviations regarding the formation of secondary inorganic aerosols (Lecœur and Seigneur 2013, Tombette and Sportisse 2007). Secondary inorganic PM is shown to represent a considerable fraction of the overall PM mass (Lecœur and Seigneur 2013). At the same time, $PM_{2.5}$ causes the highest share of quantifiable health damages in this thesis (cf. section 7.3.1). Therefore, it is important to conduct further research on the formation mechanisms of secondary PM and to improve atmospheric models like Polair3D accordingly.

Health impact assessment

A crucial element in health impact assessment are the concentration-response functions that describe the risk increase of developing a disease or dying prematurely per increase in ambient air pollutant concentration. Mainly related to this, some limitations and associated research opportunities shall be discussed:

- Generally, when establishing the relation between concentration change and health risk increase, different functional forms can be used:

  - In the EU context, a linear relationship between concentration and risk change is assumed (Holland 2014b), as adopted in this thesis and conceptualised in equation 3.3;

  - In the US context, a log-linear relationship is commonly assumed (U.S. EPA 2012b), however being approximately linear under common European background conditions;

  - A recently published approach for mortality impact assessment uses so-called integrated exposure-response functions, including effect thresholds at very low concentrations and an increasing saturation at very high concentrations (Burnett et al. 2014). These curves are approximately linear in the concentration ranges typically found under European background conditions.

  To sum up, under current European background conditions, the alternative approaches do not deviate much from the linear relationship assumed within this thesis (cf. equation 3.3). If future conditions shall be assessed, characterised by lower pollution concentration levels, it appears justified to test the implications of alternative functional forms.

- As mentioned in section 6.1.3, concentration-response function are often derived from epidemiological studies, based on long-term population cohort observations. Under ideal settings, the characteristics between the population for which a health impact assessment is carried out and the population observed during epidemiological studies, which underpin the health impact assessment, should be similar. This concerns elements such as gender and age distribution, health status, ambient pollutant concentration levels, exposure patterns etc. However, as epidemiological studies are costly and not universally available, results are often transferred from one context to another. Depending on the level of dissimilarity between population characteristics, the error induced by this transfer can be considerable (Anenberg et al. 2016, Ready et al. 2004). In the particular case of this thesis, the transfer of concentration-response functions is less of an issue given that only parameters recommended for use in Europe by the WHO (2013a) were used. These have been either derived directly for the European context or transferred from the US, where population characteristics are assumed to be similar.

- In the social CBA conducted in section 7.3.1, health impacts based on scenario calculations for one exemplary year are largely extrapolated statically into the future. The only dynamic element is the use of a life table approach for the assessment of future mortality impacts due to exposure during one year (cf. section 6.1.3.2). This static approach represents a simplification of reality, given that background conditions and assessment parameters are evolving over time, e.g. concerning pollution levels, health status and associated baseline rates, life expectancy, concentration-response functions etc. Proceeding towards a more dynamic social CBA, particularly when carrying out prospective studies, thus represents an interesting research opportunity. However, given an inherently uncertain future, it should be based on solid assumptions regarding future developments. Carrying out sensitivity analyses appears important.

- Evidence about the relative toxicity of different particles is ambiguous (Krall et al. 2013, Levy et al. 2012, Reiss et al. 2007, Stanek et al. 2011, WHO 2013b). The WHO recommends equal toxicity for all particle components (WHO 2013a), as adopted in the case studies of chapter 7 in this thesis. At the same time, different particle components are acknowledged to trigger different health effects (WHO 2013b), showing a need for further research.

- Another research opportunity is the cause-specific assessment of mortality impacts. According to latest WHO recommendations, specific causes of death are to be assessed in a sensitivity analysis only, while all-cause mortality assessment remains the default approach. This is mainly due to data availability and reliability reasons (WHO 2013a). Yet, related sources state that using cause-specific risk functions and baseline rates would yield more accurate results (Burnett et al. 2014, Miller et al. 2011, WHO 2013b). In an exploratory study, Amann and Schöpp (2011) found damage costs to generally increase when assessing cause-specific mortality impacts. However, own exemplary estimates for France (not included in this thesis) yield opposite results: combining risk slopes from the Global Burden of Disease study (Institute for Health Metrics and Evaluation 2014), based on WHO (2013a), with country-specific background rates (WHO 2014) yields lower mortality impacts per pollution increment than an all-cause mortality risk approach. This is mainly due to the substantially lower baseline rates of the four specific cause categories compared to the all-cause mortality baseline rate. Strongly depending on the context and in particular on the choice of baseline rates and risk slopes, the implications of a cause-specific mortality valuation approach thus deserve more research.

- An ethical concern is related to using an approach based on life years lost for the assessment of mortality impacts due to long-term pollution exposure, as already discussed in section 6.1.3. The fact of implicitly assigning fewer life years lost to the death of elderly people is questioned, as it could, under certain circumstances, lead to age discrimination (OECD 2016).

- Finally, a more regionalised health impact assessment at European scale could be envisaged. At the European domain, EU-wide average baseline rates, age group fractions and concentration-response functions were used. Country-specific parameters were only used at the French and Île-de-France domain (cf. section 6.1.3). A potential improvement would be to proceed towards a more regionalised assessment at European level by using country-specific data (IOM 2011, WHO 2013a). Higher baseline rates, for instance, imply elevated mortality impacts due to $PM_{2.5}$ for Eastern European compared to Central or Western European residents (Miller et al. 2011). However, such an approach may suggest region-specific emission reduction strategies. Less developed countries, often suffering from higher pollution levels, therefore risk to be confronted with substantially increased emission reduction requirements and ultimately higher costs. Although being economically efficient, a more regionalised health impact assessment may thus raise associated equity concerns.

Monetary valuation

The most critical parameter in the monetary valuation of health impacts concerns the assessment of premature deaths and the associated life years lost. Apart from the discussion on the preferred approach and the preferred central estimate (cf. section 8.1.1), more fundamental ethical objections exist.

For instance, when using surveys to value what people are willing to spend in order to avoid the risk of dying prematurely due to air pollution or other causes, the available income plays a key role (Walz 2009). Inevitably, this leads to large discrepancies in the valuation of human life between developed and developing countries, which gives rise to criticism. For this thesis, however, this is less of a concern, given that the same valuation parameter for mortality impacts is applied over the whole European assessment domain.

Moreover, using surveys for the valuation of intangible goods is a major topic of scientific debate, given that these surveys can only be conducted for given population samples with specific characteristics. Yet, results from such surveys are subsequently applied to whole populations. Moreover, surveys are itself subject to biases and constraints (Bateman et al. 2002, Pearce et al. 2006, Turner 2007).

These constraints, the differing recommendations at European and US level and the high weight of mortality-related health damage costs clearly point to a need for further research regarding the monetary valuation of mortality impacts.

8.4.2 Private cost assessment

Regarding the limitations of the private cost assessment, two methodological issues and one assumption underlying the private cost assessment of emission control measures shall be discussed here.

Generally, even though the TFTEI methodology used for cost characterisation (cf. section 6.2) is relatively detailed, it cannot replace a truly site-specific cost assessment, carried out by the operator of an individual emission source himself. Especially in the case of retrofitting emission control measures, technical constraints and other local specificities can lead to considerably different costs than the ones estimated in this thesis and used in the social CBA in section 7.3.1.

As another methodological constraint, a relatively simple cost characterisation was carried out in section 6.2. In particular, the input data used for the cost characterisation, e.g.

energy or resource costs, was assumed to remain constant over the project lifetime. Although the impact of this simplification on the outcome of the social CBA presented in section 7.3.1 is expected to be limited, exploring the effects of a more dynamic cost assessment represents an interesting research opportunity.

A critical assumption concerns the costs of switching to a very low-sulphur fuel, as explored in section 6.2. This cost is likely overestimated given that the source of the cost estimate is somewhat outdated and does neither reflect recent market developments (period of low oil prices in general), nor legislative changes (e.g. stricter European regulations regarding the sulphur content of marine fuels, cf. Directive 2012/33/EU). Although the impact of using a different cost for fuel switching on the social CBA outcome in section 7.3.1 is once again expected to be limited, an updated cost estimate would nonetheless lead to more robust conclusions.

8.4.3 Social CBA of emission control measures

The limitations discussed hereafter are mainly general concerns regarding the social CBA methodology, complemented by some specific limitations concerning the social CBA of emission control measures carried out in this thesis.

Limitations underlying the neoclassical economic approach to dealing with environmental problems, also applying to social CBA, have already been discussed in section 3.1.4. These include most notably concerns related to inter- and intra-generational equity, the concept of weak sustainability and the anthropocentric viewpoint taken in environmental-economic valuation studies.

Another critique is related to the strong focus on efficiency and the binary decision criterion of (social) CBA. This allows to consider only those effects for decision-making that can be assigned a monetary value, potentially missing other important tangible or intangible impacts (cf. section 8.2.3). It is argued that CBA should therefore be used as a decision support, complemented by further relevant information (Boardman et al. 2006, Turner 2007).

A major critique, also evident through the results presented in section 8.2.2, is the risk of biased assessments due to selective methodological choices according to the user's preferences. One way to address this issue is by transparently describing the complete input data into the social CBA as well as using latest scientific recommendations, as demonstrated in this thesis. Standardised methodological guidelines at national or international level are therefore important prerequisites for a harmonised approach to social CBA. No such guidelines for site-specific social CBA of emission control measures at European level exist yet, although some activities are currently ongoing[32]. Another factor is a better awareness on the potential influencing factors to which this thesis brought new insights.

Comprehensive uncertainty assessments improve the robustness of social CBA results for use in public and private decision-making. As discussed in section 8.2.1, existing probabilistic uncertainty analyses at aggregate level partly ignore elements related to atmospheric modelling. Moreover, they sometimes include subjective choices regarding probability estimates. An interesting research opportunity would thus be to strive for a more comprehensive probabilistic uncertainty assessment of health damage costs in Europe for use in policy assessments. To this end, existing evidence from different disciplines, particularly air quality modelling, should be combined and comprehensively reviewed.

Due to considerable expertise and resources needed, such a comprehensive probabilistic uncertainty assessment appears less feasible for private decision-making, e.g. the social CBA conducted in chapter 7 of this thesis. Increasing the modelling precision and performing sensitivity analyses are seen as more useful than an exhaustive probabilistic uncertainty assessment in a private decision-making context.

8.5 Using the social CBA methodology in the business context: methodological recommendations

Environmental regulations aiming at the internalisation of environmental and health damages push the private sector to consider related impacts, potentially even to quantitatively assess these impacts. As this thesis shows, there are opportunities to avoid an excessive cost burden in cases where overall welfare is expected to be reduced by too stringent policies (cf. section 2.4.6). To this end, the social CBA methodology needs to be transferred from the aggregate policy level to the level of an individual emission source, as demonstrated throughout this thesis. Applying social CBA at the level of an individual site bears several methodological challenges, e.g. related to choosing the appropriate methods, the appropriate modelling resolution and the treatment of uncertainty.

[32] Cf. http://ec.europa.eu/environment/industry/stationary/studies.htm, last accessed: 2017-05-18

Based on the insights generated within this thesis, this section aims at providing methodological guidance for the private sector concerning the use of social CBA for the assessment of emission control measures, notably concerning health damage cost assessment. Private cost characterisation methods are not addressed here, as they are usually well established in the private sector.

How to choose an appropriate method and implementation for health damage cost assessment?

Operators aiming to perform an environmental-economic health damage cost assessment face several possible methods, each with related strengths and weaknesses:

Option 1: The most simple and straightforward way, particularly in the context of the EU's Industrial Emission Directive, is the use of national average unit damage cost factors (cf. section 7.3.2) from the Integrated Pollution Prevention and Control Reference Document on Economics and Cross-Media Effects (European Commission 2006a). As a strength, these values stem from an official European Commission guidance document, include sensitivity results concerning mortality impact assessment, and are readily available for the most common classical air pollutants. As a drawback (cf. also section 8.1.3), these average damage cost factors are highly uncertain given that they fail to account for emission source characteristics such as the specific place and height of release and given that the underlying assessment parameters are out of date.

Option 2: As a next best solution, unit damage cost factors based on parameterised atmospheric modelling, e.g. as implemented in the EcoSenseWeb model can be used (cf. section 7.3.2 and Figure 8.1). Regarding strengths, particularly when assessing effects from large combustion plants, specific results are available for a high height of release and for different sub regions in larger European countries such as France or Germany. Moreover, different background emission inventory data and different meteorological data can be used to generate results. Nevertheless, results per substance emitted are not readily available and need to be extracted using the EcoSenseWeb model, as recently done for Germany by van der Kamp et al. (forthcoming). Taking a business perspective, the most relevant weaknesses are the limited spatial and temporal modelling resolution, as well as partly outdated input data to atmospheric modelling and health impact assessment (cf. section 5.5.2). This lack in resolution is particularly critical when effect thresholds shall be accounted for. Finally, no option to assess direct NO_2-related health impacts is currently available.

Option 3: The most sophisticated option for a health damage cost assessment uses specific atmospheric modelling and case specific input data and parameters to assess the impacts of emission scenarios, such as developed and presented throughout chapters 6 and 7 of this thesis. Regarding strengths, such an approach enables the user to: account for temporal variations in emission patterns; choose a spatial modelling resolution that is adapted to the case (e.g. high resolution in case of steep population density gradients near the emission source); increase the modelling precision for a more robust consideration of non-linear atmospheric processes; conduct a detailed assessment of uncertainty and variability underlying the health damage cost assessment. Regarding the limitations and drawbacks (beyond those discussed in section 8.4), the most notable are: the high degree of freedom in choosing modelling features, input data, and further methodological assessment principles. Inevitably, this involves the risk of biased assessments, where users would influence the outcome according to their personal preferences. This risk can be confronted through full methodological transparency, using publicly available data and models as far as available, and relying on the latest scientific state-of-the-art for parameter choices. Moreover, sensitivity analysis of key parameter or modelling choices help showing the range of plausible results. A rather practical and resource-oriented drawback is the high complexity involved in modelling atmospheric processes and chemistry by Eulerian models, such as used in this thesis. Expert knowledge and substantial computational resources[33] are needed for such a task that may only be warranted where the stakes are sufficiently high. Moreover, as a technical constraint, a site-specific approach using Eulerian models at larger geographical scale requires a minimum emission intensity in order to generate a sufficiently visible and robust signal in terms of concentration changes, as discussed, for instance, by Brandt et al. (2012).

As a conclusion from the above presented options, no “one-size fits all” approach to health damage cost assessment exists. A higher precision in the results comes at the expense of higher resource costs and an increasing risk of biased choices. Therefore, the preferred approach depends on the specific question to be addressed, as outlined in the following:

- In case of a simple screening exercise, i.e. in order to get an idea about the order of magnitude of environmental and health damage costs related to a given emission scenario or a given emission source, it may be sufficient to rely on the simplified approach mentioned above (option 1);

[33] The simulations for the case studies carried out within this thesis took several months on a high-speed parallel computer to complete.

- In case that a higher precision is needed, for instance in the context of regulatory requirements, a parameterised or even specific modelling are generally to be preferred. Under the following conditions, a parameterised modelling (e.g. using a 50 km grid cell resolution as in the case of the EcoSenseWeb model) can be considered sufficiently robust (option 2):

    - where local results (i.e. over an area of around 50 km around the emission source) are of minor importance and the focus is on regionalised results (national or European domain);

    - where population densities are relatively even around the emission source (i.e. over an area extending to 50 km or larger around the emission source);

    - where the temporal emission pattern of the emission source is relatively constant throughout the year (e.g. in the case of base load power plants);

    - where effect thresholds play a minor role.

- On the contrary, if these conditions are reversed, a site-specific modelling appears recommendable (option 3). Beyond that, a site-specific modelling is also justified where the stakes (political or financial) are very high.

When aiming to increase the spatial modelling resolution while limiting resource requirements, a pragmatic solution is to combine existing parameterised modelling results with new specific modelling results:

- A specific, more highly resolved modelling is used for the local or national scale, enabling to capture spatial and temporal variability and reducing related uncertainties;

- Results at the larger scale, e.g. European level, are derived on the basis of already existing, parameterised modelling results at a lower resolution.

When aiming to increase the temporal modelling resolution at moderate resource requirements, parameterised model results could be enhanced by accounting for seasonal variations. Given that the formation of secondary PM and ozone is known to be season-dependent (Kaneyasu et al. 1995), such an approach would improve the precision of parameterised models while at the same time limiting computational demands. The choice of the appropriate modelling resolution remains crucial and strongly depends on the case to be analysed. Generally, a higher need for precision at local level and higher stakes justify a higher modelling resolution.

The methodological recommendations given above are summarised by means of a flowchart in Figure 8.2.

How to deal with uncertainty beyond choosing the assessment approach?

Globally, it is important to take a step-wise approach to reducing uncertainty, given that uncertainties are propagated throughout the assessment chain of the impact pathway approach. Reducing uncertainties at an early stage, i.e. starting with atmospheric modelling, is therefore critical.

The different types of uncertainty underlying health damage costs (cf. section 3.4.3) can be reduced as follows:

- Variability is inherent in the modelled systems and, as such, cannot be avoided. However, spatial or temporal variability can be captured in the modelling approach through an increase in resolution of input data and spatial or temporal coverage;
- Parameter uncertainty is best reduced by complying with the scientific state-of-the-art, such as followed in this thesis. Concretely, this implies using latest WHO (2013a) recommendations for health impact assessment at EU level and for further input parameters, e.g. discount rates. Given that official guidance at EU level is currently lacking for monetary valuation, sensitivity analysis regarding the most influential parameters (i.e. valuation of mortality impacts) should be conducted;
- Model uncertainty is best reduced by selecting models that are adequate for the specific assessment context (e.g. in terms of scope and complexity) and by using input data that is as representative as possible for the case to be assessed, e.g. in terms of temporal and spatial resolution and in terms of up-to-dateness. Where feasible, modelled concentrations should be compared with ambient monitoring data in order to assess the performance of the atmospheric modelling. Advanced probabilistic uncertainty assessment methods can be used to assess the combined effect of parameter and model uncertainty;
- Decision-rule uncertainty is best confronted by the use of sensitivity analysis.

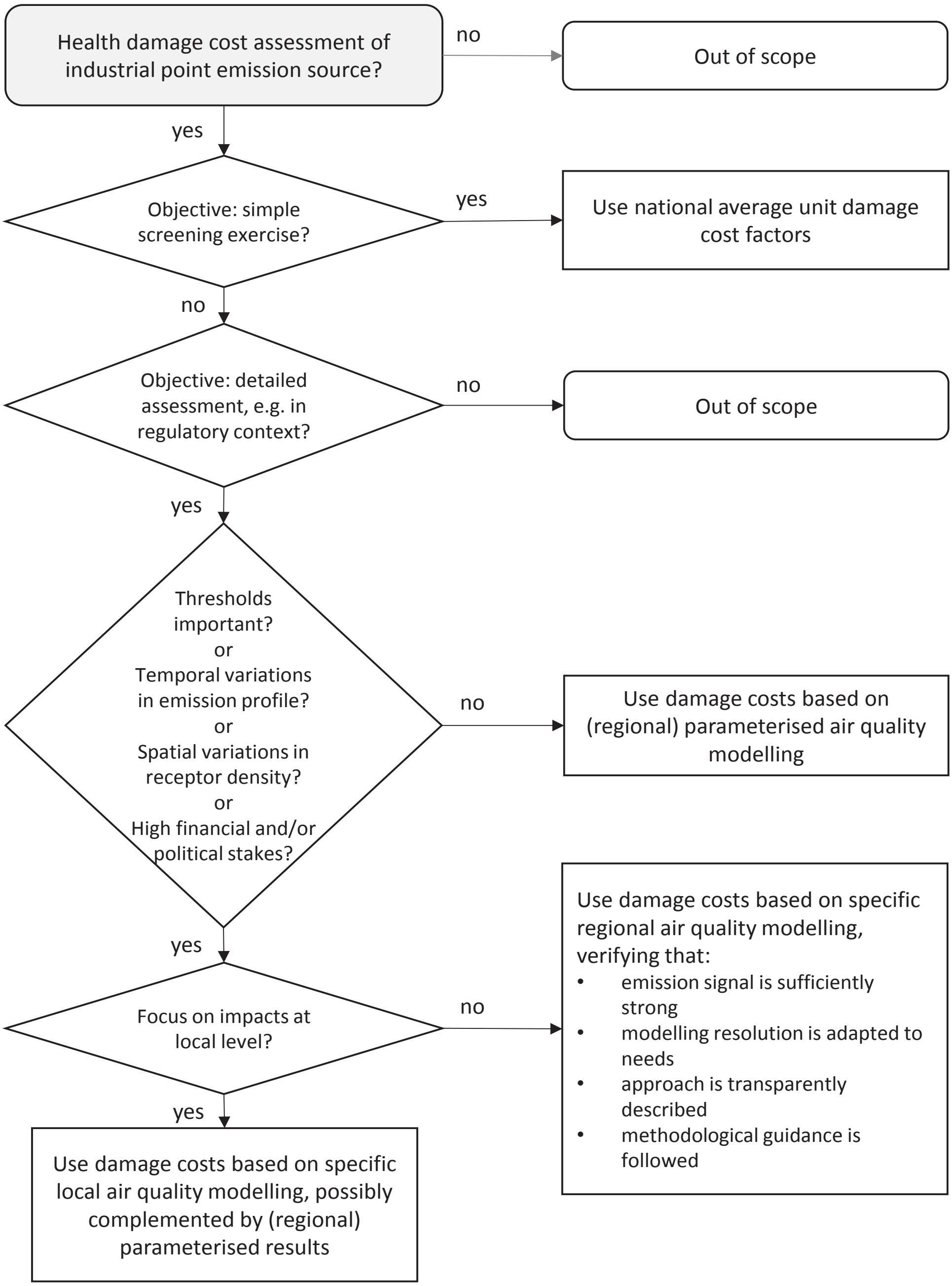


Figure 8.2: Flowchart representing the choice of the basic methodological approach to be used for health damage cost assessment at the level of an industrial point emission source (own conception)

8.6 Summary of chapter 8

This chapter starts by discussing the case studies of chapter 7 in view of underlying causal mechanisms, relevance and robustness of results.

Uncertainty underlying health damage costs is dealt with in various ways. For a quantitative view on uncertainty, the sensitivity of health damage costs with regard to modelling features and methodological choices is summarised (cf. Table 8.1). Among the various case studies presented in this thesis, emission source characteristics, spatial and temporal modelling resolution, background meteorology, and the approach for mortality impact assessment are shown to be most influential. Due to the proximity of a metropolitan area, NO_2-related impacts at local level are likewise found to considerably contribute to overall health damage costs, however depending crucially on the concentration threshold level and modelling resolution.

Owing to the scope of this thesis, certain elements remain unquantified that have the potential to affect the conclusion of the social cost-benefit analysis of emission control measures. These elements are qualitatively reviewed (cf. Table 8.2). Important omissions concern dynamic aspects within the social cost-benefit analysis (direction of effect unclear), ecosystem-related impacts (increase in societal benefits), and issues related to security of supply (potential increase in societal costs).

The impact of uncertainties on decision-making depends on the context, type, and scope of the assessment. Whilst the consequences of uncertainty are argued to be less critical in policy decision-making than typically assumed, the impact at the level of individual industrial emission sources is higher. Yet, the tolerable level of uncertainty depends strongly on the decision context and related political or financial stakes.

Extending beyond uncertainty, a critical appraisal of the methodology developed in this thesis reflects further limitations. Many of these are related to health impact assessment and monetary valuation of intangible goods, mortality impacts in particular. Moreover, global constraints of social cost-benefit analysis are discussed, such as the narrow focus on monetarily quantifiable effects or the high degree of freedom in methodological choices.

Building upon the above findings, the chapter concludes by giving methodological recommendations. These should facilitate the use of social cost-benefit analysis and health damage cost assessment in particular at the level of an individual emission source. Gains in precision through a specific modelling come at the expense of higher resource requirements and a risk of biased decision-making. Resource requirements can be limited by relying on existing modelling results, complemented with specific modelling results for particular applications, e.g. in the presence of large spatial or temporal variations. Biased decision-making is best avoided through a high level of transparency and following guidance from official parties, where available. In addition to this, sensitivity analysis is highly recommended for more robust decision-making, serving to confront some key constraints of social cost-benefit analysis.

9 Summary and outlook

Recent environmental policy developments in Europe open up the possibility of using social cost-benefit analysis (CBA) to assess emission control measures at the level of an industrial point emission source. One typical aim of social CBA in the given context is to assess whether investments into emission control measures are disproportionate by comparing private costs with societal benefits (essentially avoided health damage costs). This method is well established for aggregate policy assessments, e.g. at sectorial or European level. However, as discussed in this thesis (cf. sections 3.5 and 5.5.2), most existing assessment models fail to account for temporal and spatial variations related to atmospheric modelling and exposure assessment. As a result, they are not deemed appropriate for decision support at the level of an individual point emission source.

Against this background, this thesis set out the following objectives:

1) Extend the existing scientific methods for the assessment of human health damage costs caused by atmospheric emissions from fossil fuel power plants in order to cope with temporal variability in operation profiles and spatial variability in population densities;

2) Transfer the methodology for social CBA of emission control measures from the public policy (aggregate level) to the private business context (site level);

3) Derive methodological recommendations accounting for uncertainty in health damage costs.

All three objectives were achieved through the research conducted in this thesis, further summarised below.

Regarding objective 1, a new modelling framework for health damage cost assessment was developed, as transparently and comprehensively described in section 6.1. It considers the latest scientific state-of-the-art and follows the stages of the impact pathway approach, i.e.:

- atmospheric modelling of emission scenarios using an advanced Eulerian chemistry transport model at three spatial domains (with respectively differing spatial modelling resolutions) and at hourly temporal resolution during one year;

- population exposure assessment based upon spatially resolved data within a geographic information system, allowing the assessment of concentration changes, caused by the different emission scenarios, in combination with population data;

- health impact assessment regarding mortality- and morbidity-related impacts, caused by human exposure towards ambient air pollutants (i.e. particulate matter ($PM_{2.5}$ and PM_{10}), ozone (O_3), and nitrogen dioxide (NO_2);

- monetary valuation, assigning tangible and intangible valuation components to the physical health effects assessed in the previous step.

Given that different power plant operation scenarios are compared in this thesis, it was decided to express the resulting health damage costs in € per MWh of electricity generated.

For the purpose of objective 2, the newly developed approach for health damage cost assessment at site level (cf. objective 1) was combined with a private cost characterisation method. The annuity method was used to determine annual costs of selected primary and secondary emission control measures at an exemplary fossil-fired peak-load power plant (cf. section 6.2). This permitted to carry out an exemplary social CBA at site-level (cf. section 7.3). The emission reduction measures considered were not found to be socially efficient according to the decision rule of social CBA. Yet, as shown in complementary analyses, switching to a simplified health damage cost assessment or using an alternative mortality impact assessment approach altered the result, making the investments socially efficient. This is a typical example of decision-rule uncertainty related to selecting an appropriate health damage cost assessment approach.

Using the newly developed approach for health damage costs, key modelling features and methodological choices were varied and the quantitative sensitivities of health damage costs with regard to single elements were summarised (Table 8.1) and discussed (cf. section 8.1).

Choices concerning spatial and temporal modelling resolution were confirmed to be important influencing factors. However, their influence depends on further, site-specific elements, particularly emission source characteristics and the population distribution. For this reason, findings regarding the sensitivity of health damage costs with regard to changes underlying atmospheric modelling cannot be generalised. Nonetheless, these findings make it possible to estimate the potential error induced by using an average modelling approach instead of a highly resolved specific modelling approach. As shown in the case studies of this thesis, this error led to at least a doubling of the resulting health damage costs, thus being crucial for decision-making.

Methodological choices in health impact assessment were likewise shown to be influential, particularly those related to quantifying mortality impacts due to long-term $PM_{2.5}$ exposure, which is the single most influential endpoint (contributing around 68% to the overall quantified damages when excluding direct NO_2-related damages, cf. section 7.3). Switching from an assessment approach based on years of life lost to an approach based on cases of premature deaths universally increased associated health damage costs at least threefold (cf. section 7.2.2).

At the local modelling domain, characterised by the presence of a large metropolitan area, NO_2-related mortality impacts were shown to contribute considerably to overall health damage costs (cf. section 7.2.1). This result, however, depends crucially on the concentration threshold, above which impacts are assumed to occur. As a consequence, the spatial atmospheric modelling resolution should be sufficiently high when aiming to properly account for threshold effects.

For the purpose of objective 3, uncertainties underlying social CBA and particularly the health damage cost assessment were comprehensively reviewed in a quantitative and qualitative way (cf. section 8.2), including but not limited to the sensitivity analyses presented above. This review also provided the basis for methodological recommendations (cf. section 8.5 and below).

While health damage costs were shown to be very sensitive towards modelling parameters and assumptions, the tolerable level of uncertainty in decision-making depends on the decision context and the associated financial or political stakes (cf. section 8.3). Moreover, the impact of uncertainties on decision-making depends on the type and scope of the assessment. In public decision-making and generally when conducting scenario comparisons, e.g. within social CBA, modelling and parameter uncertainties are argued to have a smaller influence. For decision-making concerned with assessing industrial point emission sources from a private or regulatory perspective, however, the consequences of uncertainty can be substantial. This points to the need to establish methodological guidance regarding the use of social CBA of emission control measures at the level of an individual emission source in the regulatory process. Due to the fact that private cost assessment is common practice for businesses, official guidance in this field can be limited to key parameters, e.g. sector-specific interest rates. Methodological guidance is more critically needed in view of the health damage cost assessment, also due to the ongoing methodological evolutions (cf. section 8.4). Such methodological guidance, aimed primarily at private decision-makers, was therefore provided in this thesis (cf. section 8.5) and shall be briefly summarised hereafter.

An elementary choice is related to the general approach and model for health damage cost assessment. As shown in section 7.3.2, the use of simplified approaches, based on average modelling, induces a risk of erroneous decisions. At the same time, the increase in precision and robustness enabled by a site-specific modelling needs to be weighed against an increase in complexity, resource requirements, and potential biases in results. The latter is due to the manifold methodological choices that could be used to influence the outcome according to the user's preferences. This risk is notably among the key constraints regarding the use of social CBA for decision-making (cf. section 8.4.3). Yet, where the financial or political stakes are sufficiently high and the temporal or spatial variations are expected to be important, a specific modelling approach appears justified. This is especially true for the case studied in this thesis, characterised both by variations in the emission pattern (peak load power plant) and by variations in population density (proximity of large metropolitan area). Alternative ways of assessing health damage costs with lower resource requirements are also proposed, e.g. by combining a local site-specific assessment with parameterised modelling over a larger zone.

In order to reduce uncertainty and avoid biased results, a site-specific modelling approach should be transparently documented and should follow the latest official methodological recommendations. This underlines the importance for policy makers to provide such methodological guidance. Stakeholder consultations present a possibility to gather further decision-relevant information beyond social CBA. Generally, sensitivity analysis of key parameter and assumptions should be conducted for private and political decision-making alike in order to enhance the robustness of the social CBA outcome and to provide a broader picture on potential impacts.

To conclude, even though the use of social CBA and environmental-economic assessment methods is subject to limitations (cf. section 8.4), it is the author's opinion that human wellbeing and environmental resources are served better by applying such methods than by ignoring or rejecting them.

Outlook

If welfare economic instruments such as social CBA are increasingly integrated into regulatory processes, a harmonised and robust framework for estimating damage costs is needed, allowing competent authorities to base their decisions on comparable grounds.

Regarding health-related damage costs of classical air pollutant emissions in Europe, the impact functions proposed by the WHO's HRAPIE project can currently be considered the first choice (cf. section 6.1.3). A similar harmonised effort for monetary valuation, especially with regard to valuing mortality impacts, is currently lacking. When aiming to provide a consistent methodological basis regarding atmospheric modelling, sufficiently resolved standardised modelling results at European level appear as a promising starting point (e.g. unit damage cost factors per sector, zone of release, and height of release). Depending on the spatial and temporal modelling resolution and the number of emission zones and sectors distinguished, the computational calculation effort could, however, be substantial. For decision-making at site level, it appears useful to combine such regional results with more highly-resolved local modelling results.

Several methodological developments are worth noting that are expected to lead to more robust health damage cost assessments in the future:

- Advances in computational power allow ever more refined atmospheric modelling and the associated assessment of health impacts or damages up to the global scale (Apte et al. 2015, Brauer et al. 2016, Lelieveld et al. 2015). Related to this, there is a trend to improve the quality of modelled concentration data through a combination of chemistry transport modelling, ambient monitoring data, and remote sensing data (Anenberg et al. 2016, Silver et al. 2016).

- The internet of things, characterised amongst other by the widespread use of decentralised, low-cost sensors, also enables new ways of monitoring ambient concentrations of air pollutants (Deville Cavellin et al. 2016, Holstius et al. 2014, Snyder et al. 2013). This could be useful for the validation of atmospheric modelling results and the improvement of the modelling performance. Further research efforts are nonetheless needed in order to assess the reliability and performance of such sensors, especially when used outdoor.

- When quantifying health impacts and damage costs, there is a trend to regionalised assessments. These will lead to a higher precision in results but may provoke new discussions on ethical grounds, e.g. related to a fair distribution of the burden of pollution reduction between more or less developed countries (cf. section 8.4).

- Generally, there is an ever growing scientific literature on the health impacts caused by ambient air pollution exposure (WHO 2013b). A crucial next step for increased precision would be to proceed towards a cause-specific mortality impact assessment (cf. section 8.4).

- Environmental economic survey methods for the valuation of intangible goods are likewise subject to continuous scientific improvement (Johnston et al. 2017), leading to a higher confidence in the non market-based valuation components underlying health damage costs.

Appendix

Referring to section 2.4, Table A.1 gives a comprehensive overview on the potential links between industrial emissions (pressures), states, and impacts for the case of a large fossil fuel power plant (driver). The potential environmental impacts are classified according to the so-called areas of protection:

- Human health: Upon exposure, air pollution increases the risk for human health impacts;
- Man-made environment or society: Air pollution affects building materials due to corrosion; heat discharges into rivers can impact on other heat discharging installations downstream; even though involving ecosystems, impacts on forestry and agriculture are also classified to fall into this category, as both are economic sectors;
- Ecosystems: Acid rain impacts soils and thus habitats of species; heat discharges in water bodies may lead to a lack of oxygen.

Table A.1: Overview on selected environmental pressures and related impacts caused by fossil fuel-fired power generation; adapted from Bachmann and van der Kamp (2014)

Pressures	State changes induced	Areas of protection	Examples of Impacts
Greenhouse gas (GHG) emissions	GHG concentrations in the air implying changes in, for instance: Temperature, glacier volumes, intensities and frequencies of natural hazards, sea level	Human health	Negative: Heat stress, increased prevalence of infectious diseases Positive: Less cold stress
		Man-made environment/ society	Negative: Abandoning coastal areas, dyke building, increased demand for cooling, migration of people, water shortages, social instabilities Positive: New agricultural opportunities (e.g. in Nordic countries), reduced demand for heating, fertilizer effect of CO_2 in forestry and agriculture

Pressures	State changes induced	Areas of protection	Examples of Impacts
		Ecosystems	Negative: Decline of coral reefs due to ocean acidification, sea level rise, shift in climatic zones to which ecosystems/species cannot adapt, biodiversity and habitat impacts due to water shortages and sea-level rise, crops and trees not adapted to increased temperatures
Emissions of classical air pollutants	Concentration of classical air pollutants indoor and outdoor, also implying state changes in pH of precipitation	Human health	Negative: Cardio-pulmonary morbidity and increasing mortality risk, cancers
		Man-made environment/ society	Negative: Corrosion and soiling of building materials, crop yield losses, acidification Positive: Fertilizer effect of SO_2 and NO_x on agricultural land (at low input rates)
		Ecosystems	Negative: Biodiversity and ecosystem service impacts due to eutrophication and acidification
Releases of heavy metals and other toxic substances	Concentration of heavy metals and hazardous chemicals in soils, water and air	Human health	Negative: Cardio-pulmonary morbidity, cancers…
		Man-made environment/ society	Negative: Certain areas can no longer be used for specific purposes (e.g. playgrounds for children, residential areas)
		Ecosystems	Negative: Biodiversity and ecosystem service impacts due to soil and water pollution
Heat emissions	Air temperature, water temperature	Human health	Negative: Local heat production may aggravate heat stress Positive: Local heat production may alleviate cold stress

Pressures	State changes induced	Areas of protection	Examples of Impacts
		Man-made environment/ society	Negative: Restrictions of power plants to inject cooling water into rivers due to heat inputs upstream
		Ecosystems	Negative: Biodiversity and ecosystem service impacts due to lack of oxygen
Noise (and vibrations)	Noise level	Human health	Negative: Hypertension and ischaemic heart disease, disamenity (e.g. sleep disturbances)
		Man-made environment/ society	Negative: Building material damages.
		Ecosystems	Negative: Effects on animal's physiology and behaviour

Table A.2 shows the input data requirements of the damage cost assessment model EcoSenseWeb, used throughout chapters 4 and 5.

Table A.2: Data input requirements in EcoSenseWeb for a fossil-fired power plant; based on Preiss and Klotz (2008)

Characteristics	Unit
Plant design	
Year of assessment [a)]	Year
Period of assessment (in years) [a)]	Years
Stack height	m
Stack diameter	m
Flue gas volume stream [b)]	Nm^3/h
Flue gas temperature	K
Latitude	°N
Longitude	°E

Characteristics	Unit
Full load hours per year [c]	h/a
Electricity production per year [d]	GWh/a
Emissions of classical air pollutants	
NO_x	mg/Nm^3
SO_2	mg/Nm^3
Primary particles with an aerodynamic diameter of less than 10 µm (PM_{10})	mg/Nm^3
Primary particles with an aerodynamic diameter of less than 2.5 µm ($PM_{2.5}$)	mg/Nm^3
Emissions of trace pollutants	
Cadmium (Cd)	$µg/Nm^3$
Mercury (Hg)	$µg/Nm^3$
Arsenic (As)	$µg/Nm^3$
Lead (Pb)	$µg/Nm^3$
Chromium (Cr)	$µg/Nm^3$
Chromium-VI (Cr-VI)	$µg/Nm^3$
Nickel (Ni)	$µg/Nm^3$
Dioxins and dioxin-like compounds [e]	ng/Nm^3
Emissions of greenhouse gases (during operation)	
CO_2, CH_4, N_2O	t/a
Emissions of greenhouse gases (upstream and downstream)	
CO_2, CH_4, N_2O	t
Land use change	
Land use change [f]	-

[a] Only needed for valuing greenhouse gas emissions, i.e. not relevant here

[b] Flue gas volume stream is given under standard conditions denoted by “N” in the unit (e.g. according to the German standard DIN 1343 or ISO norm 2533). These are a temperature of 0 °C and an ambient air pressure of 1013 hPa. It shall further be verified that the flue gas volume stream and the pollutant concentrations are consistent in terms of oxygen-content.

[c] EcoSenseWeb internally distributes the emissions associated with the indicated full load hours equally across the year.

[d] The parameter “electricity production per year” means the net amount of electricity delivered to the electricity grid, i.e. gross electricity production minus on-site electricity demand/consumption.

e) Input of Toxic Equivalent (TEQ) values is required (cf. (European Commission 1999a, Watkiss 2007)).

f) 22 land use types are distinguished. Land use changes are intended to be due to the construction of the power plant. However, land use changes can be specified at any step of the fuel cycle. The underlying assumptions, however, confine those land use changes to Europe; irrelevant in the context of the current work.

The following tables (Table A.3 to Table A.5) provide background statistics used for the health impact assessment described in section 6.1.3.

Table A.3: Age group fractions and corresponding years used in the health impact assessments of this thesis; sources: Île-de-France and France: (INSEE 2009c); EU-average: (WHO 2016)

	Île-de-France 2009	France 2009	EU-average 2009
all	100%	100%	100%
Infants	1.43%	1.22%	1.07%
children5to14	12.63%	12.20%	10.43%
children5to19	18.91%	18.43%	16.24%
children6to12	8.90%	8.58%	7.28%
adults18+	76.79%	78.03%	80.84%
adults20to64	61.56%	58.63%	61.17%
adults30+	59.54%	63.04%	65.52%
adults65+	12.62%	16.82%	17.34%

Table A.4: EU and French baseline rates for 2009 per health endpoint used in the health impact assessments of this thesis

Endpoint detailed	Age group	EU baseline	FR baseline	Unit (per person)	Source
all-cause natural mortality (acute)	all	0.0108	0.0080	deaths/year	MDB
cardiovascular hospital admission (excl. stroke)	adults65+	0.0578	0.0763	cases/year	HMDB

Endpoint detailed	Age group	EU baseline	FR baseline	Unit (per person)	Source
respiratory hospital admission	adults65+	0.0297	0.0268	cases/year	HMDB
minor restricted activity day (MRAD)	all	7.8	7.8	days/year	(Ostro and Rothschild 1989)
all-cause natural mortality (VSL)	adults30+	0.0146	0.0127	deaths/year	DMDB
cardiovascular hospital admission	all	0.0240	0.0228	cases/year	HFADB
respiratory hospital admission	all	0.0139	0.0128	cases/year	HFADB
net restricted activity day (netRAD)	all	19	19	days/year	(Ostro and Rothschild 1989)
work loss day (WLD)	adults20to64	9.8739	9.8739	days/year	HFADB[a]
asthma symptom day	children5to19	0.1700	0.1700	daily incidence	HRAPIE
all-cause infant mortality	infants	0.0014	0.0012	deaths/year	HFADB
bronchitis prevalence	children6to12	0.1860	0.1860	annual prevalence	HRAPIE
chronic bronchitis case	adults18+	0.0039	0.0039	cases/year	HRAPIE
bronchitis symptom days	children5to14	0.2990	0.2990	annual prevalence	HRAPIE

[a] considering only working population; EU average used for France, since latest statistics for France date from year 1988; EU-average employment rate taken from eurostat (2015b)

Synonyms and sources: MDB = mortality database, HMDB = Hospital Morbidity Database (WHO 2016); HRAPIE = Health Risks of Air Pollution In Europe (WHO 2013a)

Table A.5: Life expectancy in EU28 member states in the year 2009 and YOLL (population aged 30+) per 1-year increase of 10 µg/m³ $PM_{2.5}$ concentration (life expectancy data from HFA DB (WHO 2016); YOLL estimates based on Miller et al. (2011))

	Life expectancy at birth, year 2009	YOLL [1/(10 µg/m³)*person*a]
Austria	80.6	0.00948
Belgium	80.3	0.00961
Bulgaria	73.8	0.01287
Croatia	76.4	0.01143
Cyprus	81.3	0.00920
Czech Republic	77.5	0.01089
Denmark	80.1[a]	0.00970
Estonia	75.3	0.01201
Finland	80.2	0.00966
France[b]	81.8	0.00900
Germany	80.4	0.00956
Greece	80.3	0.00959
Hungary	74.5	0.01249
Iceland	81.9	0.00893
Ireland	80.0	0.00975
Italy	82.1	0.00888
Latvia	73.3	0.01316
Lithuania	73.2	0.01319
Luxembourg	81.6	0.00908
Malta	80.5	0.00954
Netherlands	81.0	0.00931
Poland	75.9	0.01170
Portugal	79.6	0.00991
Romania	73.6	0.01296
Slovakia	75.4	0.01195

	Life expectancy at birth, year 2009	YOLL [1/(10 µg/m³)*person*a]
Slovenia	79.5	0.00998
Spain	81.9	0.00894
Sweden	81.6	0.00906
United Kingdom	80.6	0.00950
EU (average)	79.9	0.00980

[a] 2011 data, since no 2009 data available

[b] note that detailed life table results for France are used in the impact assessment, cf. section 6.1.3.2

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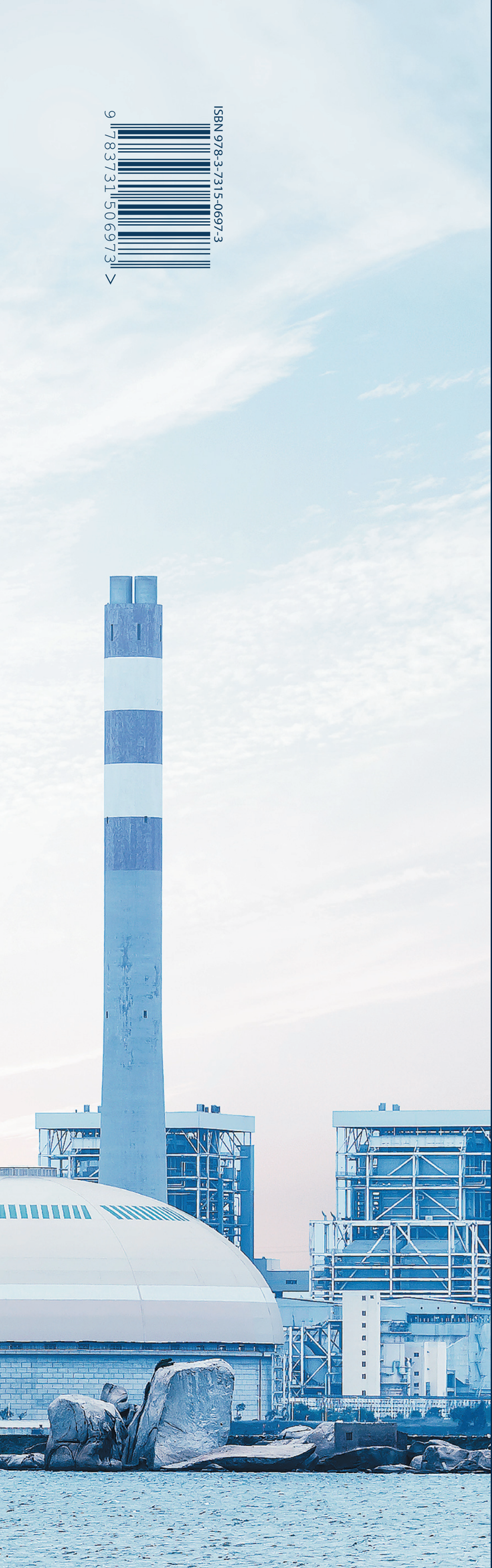

ISBN 978-3-7315-0697-3

Exposure to ambient air pollution increases the risk for humans of developing diseases as well as of dying prematurely, thus imposing costs on society. Welfare-oriented policy-making aims at integrating these costs into market prices, e.g. by reducing harmful atmospheric emissions from industrial sources. Against this background, social cost-benefit analysis (CBA) serves to identify the "optimal" level of pollution control. Social CBA related to air pollution is well developed in the public policy context. Yet, approaches at the level of industrial point emission sources are rare and existing models have important shortcomings. This work aims at enhancing economic methods for social CBA of emission control measures at site level, notably by developing a new health damage cost assessment framework and by evaluating the influence of selected methodological aspects.

Temporal and spatial modelling resolution, in conjunction with emission source characteristics and population distribution, are shown to be key site-dependent influencing factors on health damage costs. More generally, methodological choices regarding the assessment of premature mortality from long-term particulate matter exposure are particularly influential. The results underline the importance of methodological guidance from official bodies to avoid arbitrary choices in social CBA, thus facilitating the integration of health damage costs into decision-making.

Gedruckt auf FSC-zertifiziertem Papier