

EC4MACS Modelling Methodology

The ALPHA Benefit Assessment Model

European Consortium for Modelling of Air
Pollution and Climate Strategies - EC4MACS

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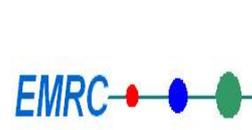
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Executive Summary

This report describes the methodology for assessing the benefits of air pollution controls using the ALPHA2 (Atmospheric Long-range Pollution Health/environment Assessment Model, version 2) and ALPHA-Riskpoll Models, and the subsequent framework for comparing quantified costs and benefits, taking account of the uncertainties that affect the analysis. The focus of the report is on regional European air pollutants, SO₂, NH₃, NO_x, fine particles and VOCs.

It is based around an earlier report produced under the Clean Air For Europe (CAFE) Programme. Significant new work is included, however. The discussion of uncertainty analysis is also better informed than in the earlier methodology reports because of the experience gained under CAFE, EC4MACS and various policy assessments. The report includes demonstration of the TUBA (Treatment of Uncertainty in Benefits Assessments) Framework.

The effects for which a detailed methodology is provided are:

- Health
- Agriculture
- Materials

Attention is also paid to methods for quantification of the costs associated with damage to ecosystems, and to integration of climate change.

Given that EC4MACS is funded by the European Commission there are frequent references in the report to assessment for European Union Member States. However, the models are set up to quantify impacts in all European countries. Indeed, the original version of the ALPHA model was developed in the 1990s to inform stakeholders throughout Europe during development of the Gothenburg Protocol under the UNECE's LRTAP Convention. In part as a result of the work performed under EC4MACS, the models have also been used to inform the revision of the Gothenburg Protocol in 2012.

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1. Introduction

1.1. Background

The benefits assessment model that forms the basis for Task 8 of EC4MACS is ALPHA2 – the Atmospheric Long-range Pollution Health and environment Assessment model, version 2. The original version of ALPHA was developed at AEA Technology in the 1990s, drawing extensively on the ExternE research programme, and was used to inform development of the EC’s Acidification Strategy, the Ozone Directive, the National Emission Ceilings Directive and the Gothenburg Protocol to the UN/ECE Convention on Long Range Transboundary Air Pollution. The starting point for EC4MACS Task 8 (ALPHA2) is the updated version developed under the Clean Air for Europe (CAFE) Programme. ALPHA2 has been developed within Microsoft Access. A second model, ALPHA-Riskpoll, was developed in Excel in the course of EC4MACS.

The word ‘benefits’ is used because most previous applications of the model have considered the benefits of new environmental policy. However, the word is used here in a very broad sense. It applies to both physical or biological benefits such as changes in health impacts (e.g. hospital admissions, reduced longevity) as well as their monetised equivalent. In some cases benefits may be negative (i.e. ‘costs’ or ‘disbenefits’).

This report describes the methodology that underpins ALPHA2 and ALPHA-Riskpoll as adopted for benefits assessment and comparison of costs and benefits under the European Commission’s LIFE+ Programme’s EC4MACS (European Consortium for Modelling of Air Pollution and Climate Strategies) Project. The report is heavily based on methods developed in earlier work for the CAFE Programme that provided input to the Thematic Strategy on Air Pollution, the CAFE Directive and the revision of the National Emission Ceilings Directive.

The original CAFE methodology report was split into three sections. The first provided an overview of the methods (Holland et al, 2005a), the second focused on health impact assessment and valuation of health damage (Hurley et al, 2005), whilst the third addressed the treatment of uncertainty in the analysis (Holland et al, 2005b). These reports were subject to extensive discussion with stakeholders. A formal peer review by independent experts was also conducted (Krupnick et al, 2005). Much of the health impact assessment is unchanged from the earlier volume 2, and so is not repeated here. There are, however, significant updates on health valuation, so further details are provided in that area.

This report is issued shortly before the conclusion of two important projects led by WHO, REVIHAAP and HRAPIE that will inform the choice of health response functions in the review of the Commission’s Thematic Strategy on Air Pollution¹. The methods for health impact assessment reported here will clearly need to be revised at the same time, though the overall modelling framework will be unchanged.

¹ <http://www.euro.who.int/en/what-we-do/health-topics/environment-and-health/air-quality/activities/evidence-on-health-aspects-of-air-pollution-to-review-eu-policies-the-revihaap-project>

1.2. Overview of Methods and Relationship to Other Models

1.2.1 The stages of the analysis

In general terms, cost-benefit analysis carried out to support policy development on air quality proceeds through the following stages:

- Scenario development, to include assessment of demand, etc., for the services of polluting activities over whatever time frame is of interest;
- Quantification of emissions under the baseline scenario;
- Quantification of emissions under scenarios involving further abatement than that specified in the baseline scenario;
- Quantification of the costs of abatement in the new scenario;
- Modelling the dispersion of emissions, including the formation of secondary pollutants (e.g. the products of reaction of SO₂, NO_x, VOC emissions in the atmosphere);
- *Estimation of exposure of receptors (people, building materials, ecosystems, etc.) that are sensitive to pollutants, combining maps of pollutant dispersion with maps of stock at risk;*
- *Application of exposure-response functions to determine various impacts, or changing levels of risk, concerning the sensitive receptors;*
- *Monetary valuation of impacts to the extent possible;*
- *Comparison of quantified costs and benefits;*
- *Consideration of the effect of uncertainties in the analysis on the balance of costs and benefits, including the omission of known and possible impacts.*

Task 8 of EC4MACS deals specifically with the second half of this process (the steps shown in *italics*). The first half of the process is carried out using other models, such as GAINS for the cost-effectiveness optimisation against specified environmental quality constraints, EMEP for dispersion modelling, PRIMES for energy sector modelling and TREMOVE for assessment of the transport sector. In practical terms the critical linkage on input to the benefits assessment is via GAINS, with no direct links being made to either TREMOVE or PRIMES. A direct link to EMEP and other dispersion models such as CHIMERE is possible, though again, GAINS outputs on air pollutant concentrations across Europe may be taken instead. The position of the benefits assessment / ALPHA2 model relative to the other EC4MACS models is shown in Figure 1.

From a purely economic perspective there is a strong argument for factoring monetised estimates of benefits into the GAINS model in order to undertake the optimisation as a direct cost-benefit analysis rather than as a cost-effectiveness assessment of meeting whatever pre-defined targets for health and environmental protection are to be considered. Cost-effectiveness analysis does, after all, only provide an indication of the cheapest way to reach a particular level of control, and does not consider whether or not that level of control is justified. This could be done as a feedback between the benefits assessment and the GAINS model. However, this step has not been taken in past work to support the development of European policy, and it is considered appropriate that under EC4MACS the comparison of costs and benefits remains outside of the optimisation process. There are several reasons for this separation:

- It highlights health and environmental improvements during debate on the level of ambition that should be considered for future policy
- It enables more transparent and comprehensive treatment of uncertainty

The first of these points is particularly important given a continued inability to monetise environmental improvements such as reduction in the area occupied by different types of ecosystem that is at risk from acidification, eutrophication or excessive concentrations of ground-level ozone. The second is important because some of the uncertainties encountered in the work make a large difference to the results of the benefits assessment (though not necessarily the conclusions drawn from CBA). It is far more transparent to account for these uncertainties outside of the GAINS model than as part of it.

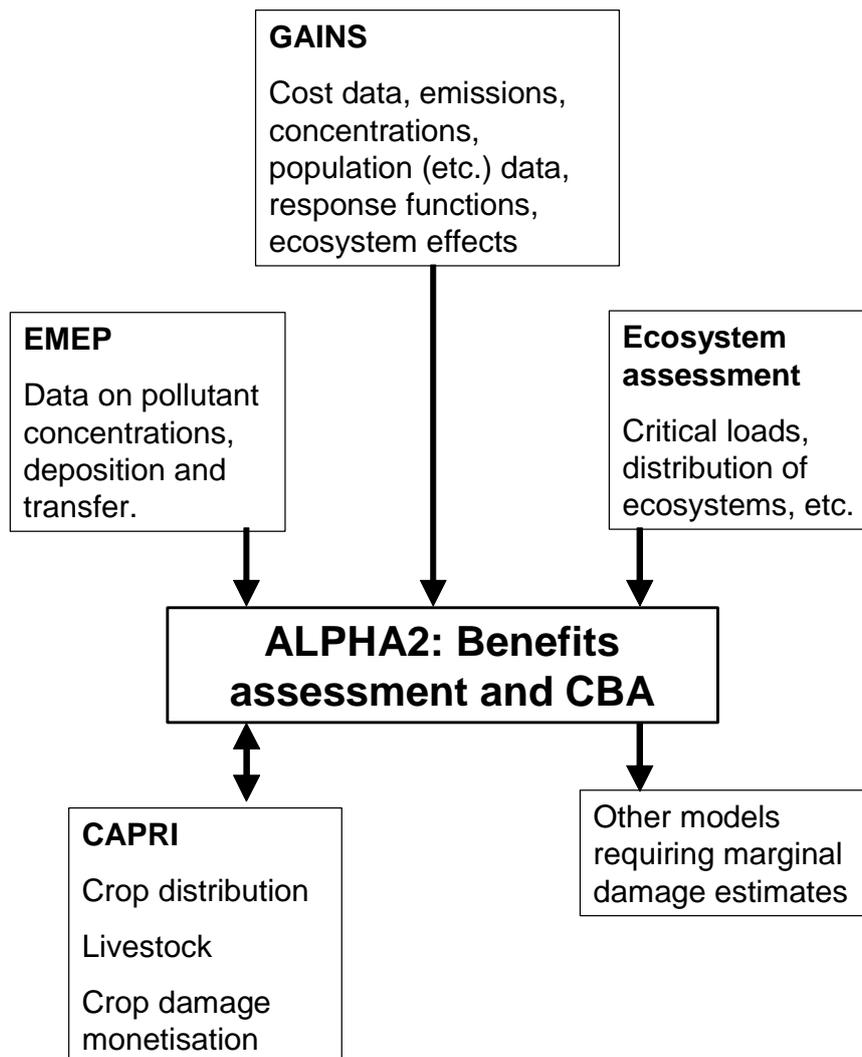


Figure 1. Links between the benefits assessment and other models.

1.2.2 Consistency of the benefits assessment and other models

The consistency of the benefits work and the RAINS/GAINS models was assessed in the earlier work under the CAFE Programme (see Appendix 3 of Holland et al, 2005a). Issues considered were:

- Currency and price year²
- Treatment of variation in values between countries
- Taxation issues
- Discounting³
- Quantification of impacts
- The basis for comparison of costs and benefits
- Treatment of uncertainty
- Input data (e.g. population data and functions and the approach for modelling mortality impacts)

The approaches used by the models were found to be the same in all areas except the treatment of uncertainty. The principal approach to uncertainty assessment in GAINS is the use of sensitivity analysis, varying certain inputs (e.g. energy scenarios) to investigate their effect on emissions, costs, etc. However, time and other constraints limit the ability of the model to take account of uncertainty using sensitivity analysis. The effects of combined uncertainties are particularly difficult to forecast. In contrast, the CBA modelling can be more flexible in its treatment of uncertainty largely because it is not tied into optimisation procedures.

Apparent inconsistency was identified between the results generated in the CAFE benefits analysis and the WHO's Global Burden of Disease (GBD) Project. This was investigated and an understanding gained of the reasons for inconsistency, which were mainly associated with the restriction of the GBD analysis to urban areas for which appropriate air quality data were available and the need of GBD to model globally rather than being focused on Europe. These differences did not apply to the main decisions made in development of methodology, for example relating to attribution of cause and effect by pollutant or to thresholds.

² For CAFE both models used year 2000 Euro as the base currency. For EC4MACS the price year has been updated to 2005 in both GAINS and the benefits assessment. Both models use a discount rate of 4% in line with the rate recommended by the European Commission.

³ Both models use the 4% discount rate adopted by the European Commission.

2. Impacts

2.1. Issues

Benefits assessment would ideally account for all of the impacts likely to be linked to the measures that will be introduced as a result of air quality policy. This includes not only direct impacts of pollutant emissions that can be modelled within the ALPHA framework but also other effects of measures that may be recommended through the cost-effectiveness modelling. The purpose of this chapter is therefore to highlight the breadth of impacts that could arise as a result of control measures that could be adopted as a result of recommendations made via the GAINS model. Consideration extends beyond the subset of measures considered by GAINS to other options, such as local controls on traffic and fiscal measures.

The chapter starts with consideration of the types of measure that could be adopted, proceeds to consideration of their impacts, and then summarises, briefly, how different groups of impact may be characterised within the modelling. In some cases this will be a full quantitative assessment, proceeding through to monetisation. In other cases it will be purely qualitative, depending on data availability.

2.2. Measures that can improve air quality

2.2.1 Types of measure

We identify the following option categories:

1. End-of-pipe controls (catalytic converters, flue gas desulphurisation (FGD), etc.)
2. Use of cleaner fuels and fuel switching
3. Promoting the use of cleaner vehicles (e.g. through establishment of low emission zones and use of fiscal incentives)
4. Increasing efficiency in the use of fuels and other inputs
5. Increasing system efficiency (e.g. through travel planning, increased adoption of CHP)
6. Generating modal shift in transport by promotion of walking and cycling, better provision of public transport, etc.
7. Use of planning controls
8. Use of fiscal incentives, such as subsidies to promote energy efficiency or green electricity, congestion charging and differential fuel taxation.

Clearly, some of these measures would be introduced primarily for reasons such as reducing congestion or increasing profit (e.g. from various efficiency measures) rather than air quality improvement per se. In the interests of developing ‘joined up’ policy it is important that opportunities offering benefits in different areas are identified.

The measures included in the GAINS model are a subset of those listed here, covering those that can reasonably be generalised at the European scale in terms of availability, state of technology, etc. These tend to be end-of-pipe controls, improvements in fuel quality, and fuel switching. GAINS thus currently excludes measures that operate on a more local scale, such as the use of congestion charging and the implementation of sustainable procurement mechanisms. Some of these measures may be more cost-effective than those contained in the GAINS model database. They may also be implemented irrespective of air quality concerns,

for example, to reduce congestion or control greenhouse gas emissions or simply to save money through more efficient use of resources.

2.2.2 Associated impacts

A generalised mapping of the categories of pollution control option given above to environmental and social burdens is given in Table 1, led from the perspective of the major regional pollutants (NO_x, NH₃, PM_{2.5}, SO₂ and VOCs). The presence of possible negatives in the table stresses that care needs to be taken to optimise abatement strategies such that they do not threaten other policy initiatives and vice-versa.

Table 1. Abatement options and general trends in associated burdens

	Regional pollutants	Greenhouse gases	Noise	Social impacts	Macroeconomic effects	Performance of SMEs
End-of-pipe controls	+	-	+/-	+/-	+/-	+/-
Use of cleaner fuels and fuel switching	+	+/-		+/-	+/-	+/-
Use of cleaner vehicles	+	+/-	+	+/-	+/-	+/-
Increased energy (etc.) efficiency	+	+		+	+	+
Increased system efficiency	+	+	+	+	+	+
Generating modal shift in transport	+	+	+	+	+/-	+/-
Use of planning controls	+	+	+	+	+/-	+/-
Use of fiscal incentives	+	+	+	+/-	+/-	+/-

Key:

‘+’ = positive effect

‘-’ = negative effect

‘+/-’ is given where direction of effect is measure-specific

Some further appreciation of the differences in impacts between the GAINS – pan-European measures and options implemented at the local scale is important. Local measures may be very cost-effective for reducing the exposure of individuals living in ‘hot-spots’, but of limited effectiveness or efficiency for protection of society as a whole. This is important when considering how legislated air quality objectives are developed.

2.3. Impacts addressed by the EC4MACS benefits assessment

The main focus of the modelling discussed here is on the impacts of the major air pollutants that are specifically of interest at the regional level – particles, SO₂, NH₃, NO_x, VOCs and NH₃. These impacts are listed in summary format below (Table 2), together with indication of whether they are:

- Addressed quantitatively, typically using the impact pathway methods through to monetisation (see Section 3.2)
- Are mainly a subject for qualitative appraisal, or

- Are considered likely to be negligible.

Table 2. Effects of the major regional air pollutants, and the extent of assessment.

Effect	Impact quantified and valued	Impact only quantified	Qualitative assessment	Comments
Health				
Primary PM, NO ₃ and SO ₄ aerosols				Care taken to avoid double counting with chronic effects. All particles considered equally harmful by mass
acute – mortality, morbidity	✓			
chronic – mortality, morbidity	✓			
infant mortality	✓			
Ozone				Less clear linkage between O ₃ and mortality than for PM _{2.5}
acute – mortality	✓			
chronic – mortality			✓	No information on possible chronic effects
acute – morbidity	✓			
chronic – morbidity			✓	
Direct effects of SO ₂			✓	Limited importance to CAFE
Direct effects of VOCs			✓	Lack of data on speciation, etc.
Direct effects of NO ₂			✓	Lack of clear information of effects at ambient levels
Social impacts			✓	Limited data availability
Altruistic effects			✓	Reliable valuation data unavailable
Agricultural production				
Direct effects of SO ₂ and NO _x			✓	Negligible according to past work
Direct effects of O ₃ on crop yield	✓	✓		
Indirect effects on livestock production		?	?	
N deposition as crop fertiliser			✓	Negligible according to past work
Visible damage to marketed produce			✓	
Interactions between pollutants, with pests and pathogens, climate...			✓	Exposure-response data unavailable
Acidification/liming			✓	Negligible according to past work
Materials				
SO ₂ /acid effects on utilitarian buildings	✓			Lack of stock at risk inventory and valuation data
Effects on cultural assets, steel in re-inforced concrete			✓	
Effects of O ₃ on paint, rubber	✓			
Ecosystems				
Effects on biodiversity, forest production, etc. from excess O ₃ exposure		✓	✓	Valuation of ecological impacts is currently too uncertain
Effects on biodiversity, etc., from excess N deposition		✓	✓	Valuation of ecological impacts is currently too uncertain
Effects on biodiversity, etc., from excess acid deposition		✓	✓	Valuation of ecological impacts is currently too uncertain
Visibility: Change in visual range			✓	Impact of little concern in Europe.
Change in greenhouse gas emissions		✓	✓	Valuation too uncertain
Drinking water supply and quality			✓	Limited data availability

Where possible the quantification of impacts and monetary values using the impact pathway approach is preferred. However, where this is not possible the TUBA (Treatment of Uncertainty in Benefits Assessments) Framework provides a mechanism for extending the CBA so that unquantified impacts may be considered in a more systematic manner than

previously. It also provides a mechanism for increasing understanding of the impacts that can be quantified.

2.4. Impacts of measures not included in GAINS

There are no plans for a systematic quantification of effects of the mainly local measures that are not included in GAINS as part of the EC4MACS benefits assessment. These measures include options such as:

- Congestion charging
- Driver training
- Implementation of sustainability criteria to procurement
- Establishment of low emission zones
- Local infrastructure changes, e.g. to improve public transport interchanges or develop new routes
- Introduction of development controls

However, the general modelling framework developed here can be applied in isolation of GAINS and at any scale from global to local (see, for example, the report on analysis of the introduction of a Low Emission Zone in London by Watkiss et al, 2003). At its simplest level this might take the form of Table 1, though expanded in key areas, for example to show which impacts of transport are affected by a measure, rather than bulking them together. At a more sophisticated level the cell entries in the table could be expanded to give an indication of the scale of impacts, whether they are likely to be large or small, not just positive or negative. These judgements would be facilitated by the growing literature on such measures. Where very detailed information is available it may be possible to extend the analysis through to a full quantification and valuation of impacts.

3. Basic Structure of the Benefits Analysis

3.1. Pathways from emission to impact

At the start of the EC’s ExternE project in 1991 a series of diagrams were produced that sought to illustrate the main effects of air pollution on health, crops, forests and other terrestrial ecosystems, freshwater ecosystems and building materials (ExternE, 1995). One such diagram, for crops, is reproduced in Figure 2.

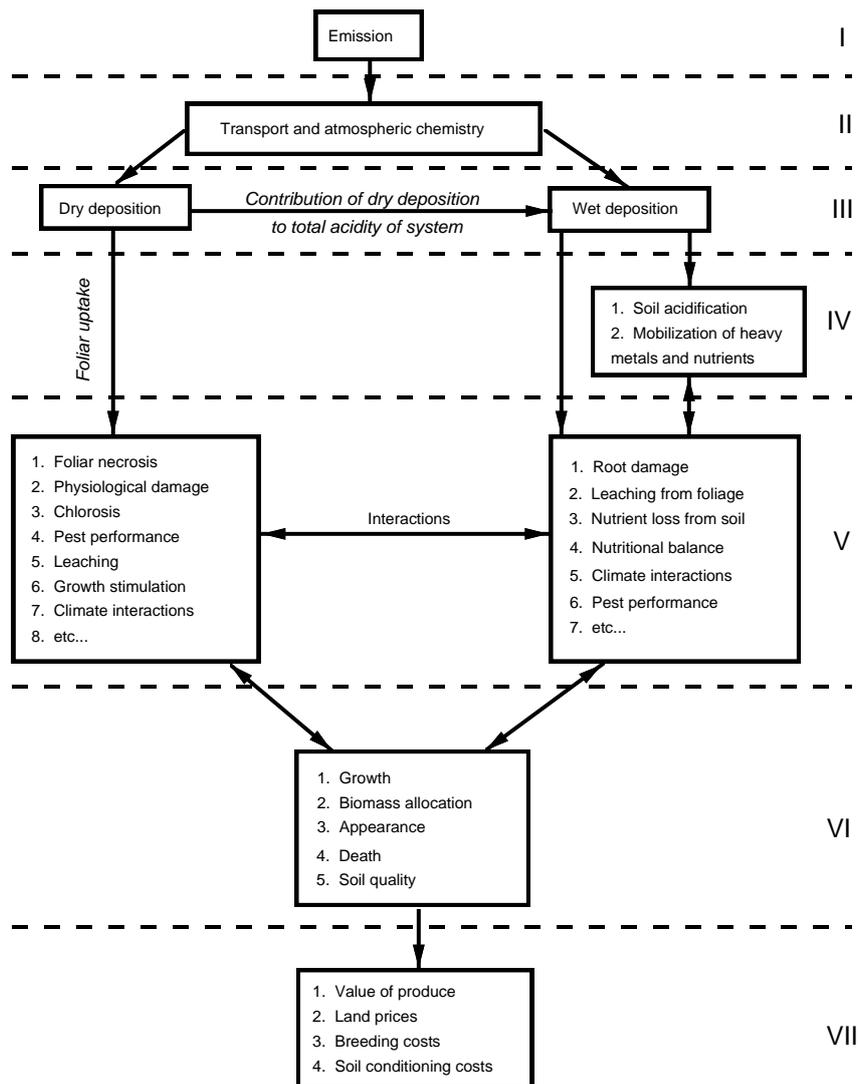


Figure 2. Detailed impact pathway to describe the way that air pollution affects crop production

These pathways contain numerous potential feedbacks and synergies that would ideally be brought together in a modelling framework. Whilst it is not possible to model at the level of complexity identified in these diagrams, they are useful for highlighting the complexity of the situation and the numerous ways in which air pollution can affect health and the environment.

3.2. Quantifying regional air pollutant impacts and monetary damages

3.2.1 Impact pathway approach

The approach taken for quantification of the benefits of air pollution emissions through to monetisation is often referred to as the ‘impact pathway approach’ (Figure 3), a logical progression from emission, through dispersion and exposure to quantification of impacts and their valuation (reflecting the overall structure of Figure 2). This approach was developed through the series of EC DG Research projects under the ExternE banner (and its predecessor, the EC/US Fuel Cycles Study) through which the approach has been widely disseminated (ExternE, 1995, 1999a). It has also been used extensively in past work on European air quality legislation (e.g. AEA Technology, 1999, 2001; Holland et al, 1999; AEA Energy and Environment, 2005a, b, c and 2006) and thus has been widely debated at a senior level.

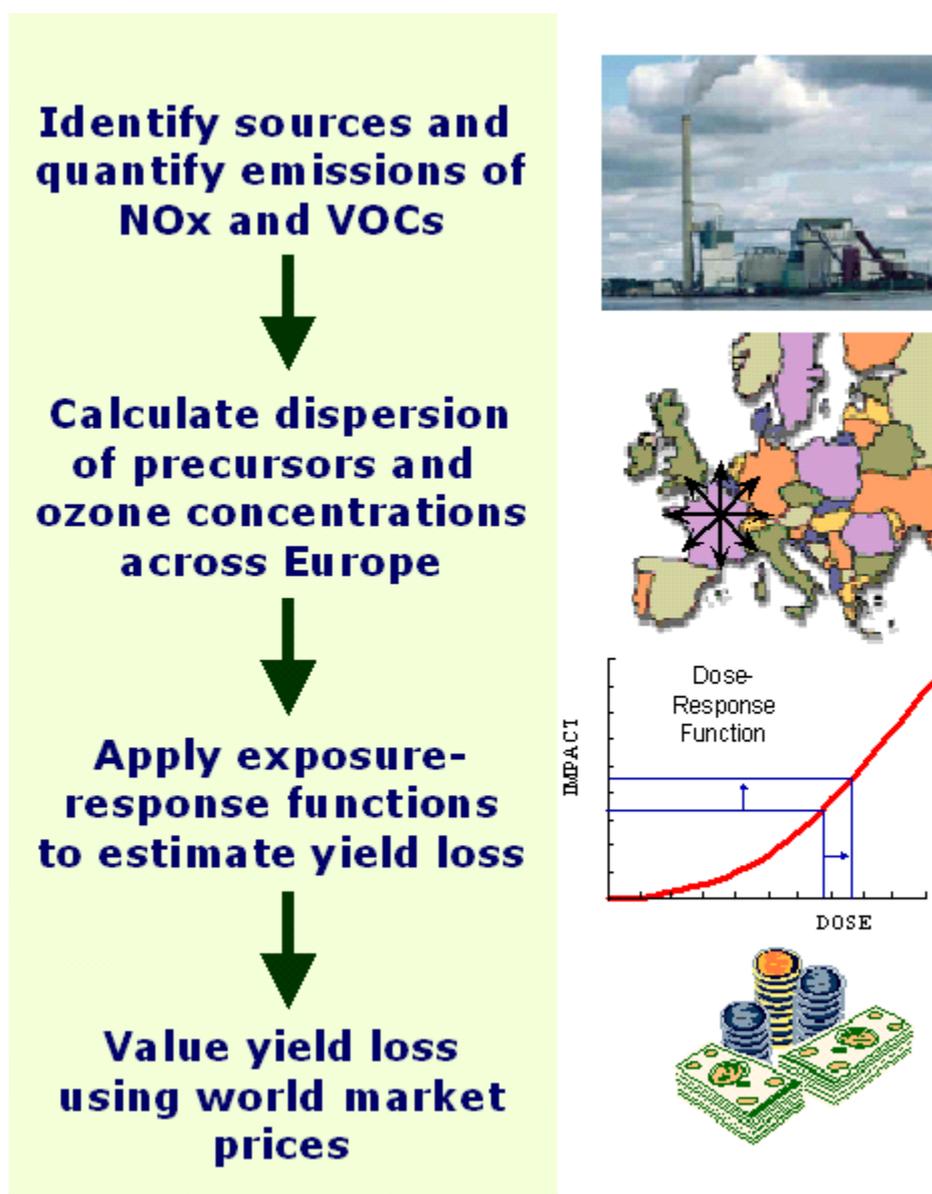


Figure 3. Illustration of the impact pathway approach taking the example of direct effects of ozone on crops

Following from Figure 3, impacts and damages under any scenario are calculated using the following general relationships:

$$impact = pollution \times stock \times response$$

$$economic\ damage = impact \times unit\ value\ of\ impact$$

Pollution may be expressed in terms of concentration or deposition. The term ‘stock at risk’ relates to the amount of sensitive material (people, ecosystems, materials, etc.) present in the modelled domain.

Calculations are normally made for each cell within a grid system: for European assessments the EMEP 50 x 50 km grid has mostly been used for the main analyses, though in 2012 there has been a shift to a finer scale. There is a possibility of additional work at finer resolutions (see Section 3.2.2). Account is also taken of elevated concentrations in urban areas. The basic form of the analysis remains the same no matter what spatial resolution is adopted for the analysis.

Although the underlying form of the above equation does not change, the precise form of the equation will vary for different types of impact. For example, the functions that describe materials damage from acidic deposition require consideration of climatic variables (such as relative humidity) and need to account for several pollutants simultaneously. Similarly, health functions require additional information on the incidence of ill health (death rates, hospital admissions, etc.).

For any type of receptor it is necessary to implement a number of these impact pathways to generate overall benefits. So, for example, in the case of impacts of ozone on crop yield, it is necessary to consider, separately, impacts on a series of different crops, each of which differs in sensitivity. For health assessment it is necessary to quantify across a series of different effects for different parts of the population to understand the overall impact of air pollution on the population.

The final stage, valuation, is generally performed from the perspective of ‘willingness to pay’ (WTP). For some effects, such as damage to crops, or to buildings of little or no cultural merit this can be done using appropriate market data. Some elements of the valuation of health impacts can also be quantified from ‘market’ data (e.g. the cost of medicines and care), though other elements such as willingness to pay to avoid being ill in the first place are clearly not quantifiable from such sources. In such cases alternative methods are necessary for the quantification, such as the use of contingent valuation (for discussion of this and other valuation techniques see ExternE, 1999a; EAHEAP, 2000). Where impacts arise in the future it is necessary to discount monetised values (but not the impacts themselves).

Worked examples of the calculations made using impact pathway analysis of health endpoints are shown in Appendix 2.

3.2.2 Quantification of benefits at different geographic scales

There is a need to quantify effects not only at the European scale on the EMEP 50 x 50 km grid and the finer EMEP grid introduced in 2012, but also at more local scales, to enable investigation of measures for addressing air quality problems in cities.

To some extent this is already factored into the Europe-wide GAINS analysis, through the use of a factor to elevate concentrations in urban areas in line with the results of the CITYDELTA project. However, it may be envisaged that the analysis will be carried out for individual cities for which air quality data are available at a much finer resolution, based on results of monitoring and urban-scale modelling.

Should analysis be required for individual cities, the main analytical complexity would be to ensure that data on stock at risk (principally population, and relevant fractions of it) can be described on the same grid system as pollutant concentration data. This should not represent a significant problem as detailed maps of population are now available for many cities, given that health impacts will dominate urban assessments.

For most analysis, the response functions and valuations used for quantification at the local scale will be the same as those used for the pan-European assessment described above. However, for some cities in which epidemiological studies have been performed it may be appropriate to also use functions taken from the data collected locally for comparison with the standard function set. We would recommend using both sets (i.e. not just the local functions) in order to build understanding of the differences that arise.

For materials damage there may be some merit as part of a local scale analysis in quantifying deterioration rates for specific buildings to illustrate the extent of problems for cultural heritage through comparison with the ‘acceptable rates’ of damage recommended by ICP Materials. As noted elsewhere in this report, monetisation of damage to cultural heritage will not be possible, given the lack of information on stock at risk, concerning not just the number of buildings but also descriptions of the materials from which they are made, and a lack of data on valuation of damages. However, knowledge of deterioration rates can be compared with ‘acceptable rates’.

The fact that the pollutants of prime interest to EC4MACS operate over distances extending beyond 1000 km creates an obvious difficulty. Models are designed to operate over defined domains, for example covering a city or EU Member States and this artificial restriction of geographic area clearly leads to a bias for underestimation of total impact. It is often not possible to resolve this problem by extending the range of the modelling. Instead, it is important to acknowledge that the problem exists and where possible to give an indication of the likely magnitude of underestimation.

3.3. Overview of presentation of the results of the analysis

Considerable attention has been given to the way that the results of the analysis are presented, particularly with respect to the way that uncertainties are reported. The following describes the sequence of the assessment:

1. Quantification of costs (GAINS)
2. Quantification of impacts (GAINS/CBA)
3. Monetisation of impacts where possible
4. Initial comparison of costs and benefits

5. Description of unquantified benefits
6. Further uncertainty analysis for benefits
 - a. Bias analysis
 - b. Statistical analysis
 - c. Sensitivity analysis for benefits
7. Overview of the likely effect of uncertainties on the balance of costs and benefits

Stages 1 to 3 need to be done at the level of totals for the EU27 and non-EU states and also at the level of the Member States.

Stage 4 is a simple comparison of best estimates of costs and benefits. This will be focused on the following series of questions:

- Across Europe, do quantified benefits exceed costs?
- What is the ratio of benefits to costs?
- If quantified benefits do not exceed costs, does it seem likely that underestimation of effects could have a significant impact on the balance between the two?
- For each country individually, do benefits exceed costs?
- What is the ratio of benefits to costs for each country?
- For countries where quantified benefits do not exceed costs, does it seem likely that underestimation of effects could have a significant impact on the balance between the two?

Stage 5, the description of unquantified impacts, has been developed with three objectives. The first is simply to increase understanding of impacts, whether they have been quantified or not. The second is to provide contextual information to assist in validation of quantified estimates. The third objective is to provide a mechanism whereby decision makers can develop a better understanding of the likely importance of impacts that have not been quantified, specifically, whether they are likely to be significant enough to alter the balance of costs and benefits. Only limited development of this part of the analysis was possible under the CAFE programme. A formal approach around the TUBA Framework was developed during EC4MACS.

Stage 6 addresses the uncertainties present in the analysis. The first part of the uncertainty assessment involves statistical analysis of the more important endpoints for the quantified benefits assessment, principally the health impacts. This enables description of benefits in terms not simply of a best estimate, but also, through the use of Monte Carlo analysis, the probability distribution around the best estimate.

This is combined with sensitivity analysis, addressing specific uncertainties identified during the development of the methodology. Under CAFE the most important were considered to be:

- The approach taken to mortality valuation
- Alternative views on the best estimate of the response function for effects of chronic exposure to particles on mortality
- Alternative views on the costs of abatement

The second part of the uncertainty assessment is a straightforward bias analysis, where the assessment of costs and benefits is reviewed and biases identified along with their direction. This will indicate, for both costs and benefits, whether either is likely to be systematically over-or under-estimated.

This detailed approach to quantification and description of uncertainty is brought together to provide an overview of the robustness of the initial conclusions from Stage 4 of the relationship between costs and benefits (Stage 7). This can be considered alongside a quantitative assessment of the probability that benefits will exceed costs for each sensitivity case.

4. Quantifying Health Impacts

4.1. Approach and data sources

This section provides a summary of our general approach to health impact quantification, together with recommendations for response modelling. Information on valuation of health impacts is provided in the next chapter. A more complete account is provided in Volume 2 of the CAFE methodology (Hurley et al, 2005, though the valuation data presented below have been updated). There has also been a further review of the health impact assessment data under the EC DG Research NEEDS Project (Torfs et al, 2007), though this adopted the same functions as had been agreed for use in CAFE.

The aim of this chapter is to present a methodology designed to generate an unbiased set of estimates of the effects of air pollution in the form of primary and secondary aerosols and ozone on health, along with guidance on the reliability of those estimates. The approach is designed to neither systematically over-estimate or under-estimate the health effects or their values, within the constraints of what is currently known with a sufficient level of confidence.

The approach presented here is consistent with the World Health Organisation's (WHO) "Systematic Review of Health Aspects of Air Quality in Europe" (<http://www.euro.who.int/document/e79097.pdf>) and answers to follow-up questions (<http://www.euro.who.int/document/e82790.pdf>) asked by the CAFE Steering Group. These are an excellent resource on hazard identification regarding the health effects associated with or caused by particulate matter (PM), ozone (O₃) and nitrogen dioxide (NO₂) but are not a 'tool-kit' for HIA of the effects of air pollution.

For quantification, the core recommendations for mortality related to PM and ozone are based on recommendations of WHO-CLRTAP Task Force on Health (TFH) (<http://www.unece.org/env/documents>). However, the aim of the benefits assessment is to compare costs and benefits of specific policies and so:

- (i) It is necessary where possible to proceed to valuation, paying special attention to the linkage of epidemiology and valuation for mortality; and
- (ii) It is also necessary to include other effects (morbidity, infant mortality).

For morbidity we have used a wider set of HIA sources that includes the work of ExternE and of Hurley et al (2000), Künzli and colleagues (e.g.; Künzli et al, 2000), major projects such as APHEIS, the Global Burden of Disease project, and the Benefits Assessment of the US EPA in evaluating the US Clean Air Act (Second Prospective Analysis) and, in particular, the WHO-sponsored meta-analyses of the acute effects of PM and ozone based on studies in Europe (<http://www.euro.who.int/document/e82792.pdf>). The CAFE methods also drew on the comments of the peer review (Krupnick et al, 2005), and subsequent correspondence with the peer review team in the identification of additional relevant studies.

4.2. Coverage of 'acute' and 'chronic' health effects

Most available studies examine the effects of *acute* exposure (usually known as *acute health effects* or effects of daily variations in pollution); i.e. the ways that air pollution on a given

day or adjacent days can affect the health of people on the same day or on the days immediately following – typically within one week⁴.

A limited set of studies examines the relationships between *chronic* (longer-term, possibly lifetime) exposure and health (usually known as *chronic health effects*). Here the strength of evidence is less than for acute effects, at least in terms of the number of studies published and variety of locations considered, but the estimated impacts are greater, because the effects of long-term exposure capture, at least partially, the acute effects, as well as including aspects not captured by the acute effects.

The key recommendations are presented in turn below for seven key areas of analysis:

- Chronic mortality from PM;
- Acute mortality from ozone;
- Infant mortality from PM;
- Valuation of mortality;
- Morbidity impacts from PM and ozone;
- Valuation of morbidity;
- Sensitivity analysis.

4.3. Chronic mortality from PM

Chronic effects, particularly on mortality have become the main focus for quantification of the health impacts of particulate exposure. Past CBA work building on the findings of ExternE, (e.g. AEA Technology, 1998 and the various analyses carried out for CAFE) demonstrated that these effects dominated benefits estimates made for the Gothenburg Protocol and the original NEC Directive.

- Consistent with WHO/TFH advice to GAINS and wider current practice, the assessment of chronic effects on mortality is based on the extended follow-up by Pope *et al* (2002) of the American Cancer Society (ACS) cohort.
- Following WHO/TFH, the following coefficient is adopted:

6% change in mortality hazards (95% CI 2-11%) per 10 $\mu\text{g}/\text{m}^3$ PM_{2.5}

- This is the coefficient relating mortality to average exposure throughout the follow-up period, unadjusted for the possible effects of other pollutants, notably SO₂.
- To quantify life years lost it is necessary to apply this function within life tables. Methods for this were described by Miller and Hurley (2003)
- At the time of the CAFE work it was also recommended to investigate the effect of using a lower coefficient of 4%, based on exposures measured at the start of follow-up in Pope *et al.* (2002). However, recent reviews (e.g. COMEAP, 2009) suggest that the 6% risk factor is less likely to overestimate impact than previously thought and arguments have been made for increasing the core risk factor (e.g. Ostro, 2007, reporting on the conclusions of an expert panel established by USEPA). On this basis, together with the use of a more detailed approach to uncertainty assessment than was appreciated when the original recommendation was made, the 4% sensitivity run is no longer considered necessary by the present authors.

⁴ Some analyses of acute exposure now include effects that occur up to 40 days from the relevant pollution days

The WHO answers to CAFE concluded that the burden of evidence - both theoretical and empirical - was very strongly against the view that there is a population threshold. This is to say that there is no background level that is safe, i.e. no level for which there is no increase in risks for any in the population. Following WHO advice to the GAINS analysis as well as the views of others, such as ExternE (1999a), the above function is therefore applied without threshold to anthropogenic PM. It should be noted that the analysis does not account for naturally derived particles in the atmosphere (whose concentration will not change with new policies). There is, therefore, an implicit threshold present in the benefits assessment, though not one that would make a difference when policies are assessed relative to a baseline scenario. This position was supported by a major recent Canadian study, where no evidence for a threshold was identified when considering the mortality risk faced by people living throughout Canada, including areas where pollution levels are significantly lower than those faced by most people in Europe (Crouse et al, 2012).

There is a question of how one should treat different types of particle, especially (given the types of control available for reducing PM exposure) sulphate and nitrate aerosols. There is a view that there is little or no evidence for effect of either of these aerosol species (see Rodgers, 2007 and Concawe, 2007). TFH, in contrast considered that there was no available evidence from which it would be possible to derive different response functions for different parts of the overall particle mix, and hence that all particles should be treated similarly. Ostro (2007) cites evidence for a significant role of both sulphates and nitrates, and possibly that they (or their correlates) are more toxic than generic PM_{2.5} mass:

To date, most analyses using sulfates have produced positive, and often statistically significant, associations including studies conducted in Santa Clara County, CA (Fairley, 1999), eight Canadian cities (Burnett et al, 2000), and several urban areas on the east coast and in the Midwest (Schwartz et al, 1996). In addition, in a recent effort to compare results from alternative factor analysis methods to estimate the effects of sources of fine particles, the sulfate-related factor was most consistently significant in the cities studied (Thurston et al, 2005). In one of the few studies examining nitrates, a positive and significant association was detected (Fairley, 1999). Finally, a study of mortality in nine Californian counties suggests that nitrates and sulfates each have a higher risk estimate per $\mu\text{g}/\text{m}^3$ than does generic PM_{2.5} mass (Ostro et al, 2007).

There are numerous well-conducted studies of the acute effects of PM on mortality (i.e. the response to short-term elevation of PM concentrations, rather than long-term exposures). The results from European studies were included in the WHO meta-analysis. However, to avoid double-counting, the EC4MACS analysis does not include these alongside the chronic mortality impacts.

The physical impacts for chronic mortality are quantified principally in terms of change in longevity. This is consistent with the methods proposed by

- WHO,
- The output from the GAINS model,
- Our own long-established practice under the ExternE Project series (see, e.g., ExternE, 1999a and Rabl, 2003), and
- Emerging consensus in HIA work

Given the WHO guidance, change in longevity aggregated across the population (otherwise referred to as ‘years of life lost’) is the most relevant metric for valuation. In order to value the benefits of reduced air pollution the concept of the value of a life year (VOLY), long advocated by the ExternE team, needs to be applied.

However, the peer review of the CBA methodology in 2005 pointed out that direct, credible estimates of the VOLY are lacking, that the estimates to be used in CAFE were derived computationally and that to be applied correctly, VOLYs should be age specific. Thus, while the physical impacts (reduced life years) can be derived without major difficulties, the derivation of values for these impacts has methodological problems⁵. For these reasons it was decided under CAFE to also quantify premature mortality benefits based on the cohort studies in terms of ‘attributable deaths’ (or, for pollution reductions, attributable deaths delayed) which can be valued using the better known ‘Value of Statistical Life’ (VSL) approach. A more complete discussion of these issues is given by Hurley et al (2005), but in summary, there are two possible approaches to estimating numbers of deaths (attributable cases) associated with long-term exposure to PM.

- The first approach uses life tables, because this is the methodology used to give results in terms of life expectancy. As far as we know, life tables are not used currently to estimate ‘attributable deaths’. Indeed, there are some important methodological problems in attempting to do so.
- The second approach does not use life tables but applies the change in mortality rate from the above function to the death rate. It is simple to implement and is widely used. It is, however, a simplification.

In CAFE it was considered appropriate to use the second simpler method because (i) it is simple to do and, despite its limitations, is widely-used; and (ii) the methodological problems associated with using deaths from life tables have yet to be resolved sufficiently. It is stressed that these methodological differences imply that there is no easy equivalence or ‘conversion’ between the estimates of life years and attributable deaths, even though both are derived using the same coefficient from the US cohort studies, and using the same pollution data. In particular, it is not valid to ‘convert’ between life years and deaths, from the results so generated.

The need to quantify impacts in terms of deaths as well as life years lost is dependent on views on the validity of estimates of the VOLY. The peer review by Krupnick et al (2005) recommended using both VOLY and VSL, partly to demonstrate the sensitivity, and partly in recognition of the fact that different users would advocate different approaches. In practical terms the choice made little difference to the conclusions reached under CAFE on scenario choice for the Thematic Strategy on Air Pollution. Taking other impacts into account, use of the VSL increased damage estimates by around a factor 2 compared to the case when the VOLY was used. However, for most scenarios this had a negligible effect on the probability of benefits exceeding costs for policy relevant scenarios.

Two potentially important biases affect the analysis:

1. The assumption of no time-lag between changes in pollution and consequent changes in death rates. This is the approach taken by GAINS and, for consistency, is adopted also here.

⁵ Since 2005 new work has been published on the VOLY that avoids some of these problems (Desaigues, 2011). However, there is still not a substantial and consistent body of literature on quantification of the VOLY so the original concerns remain to some extent, justifying retention of the VSL.

- The assumption that all types of particle are equally harmful per unit mass. This is an extremely important issue for policy, though any attempt at quantification will be speculative at the present time given a lack of empirical evidence on which to base differentiation.

In CAFE, the estimation of life years lost for all European countries was based on a life table analysis using data for the population of England and Wales. Data for some other (western) European countries was also obtained and suggested very similar results. However, further work reported by Hurley et al (2011) using life tables for newer EU Member States, and more particularly, for Russia found considerable variation between countries. In considering how to account for this without life tables for every country, a relationship between life years lost per unit exposure and life expectancy was observed (Figure 4). This was subsequently implemented to the benefit assessment models to take account of differences in life expectancy between countries (relative to England and Wales). It was also adjusted to account for changes in life expectancy in future years.

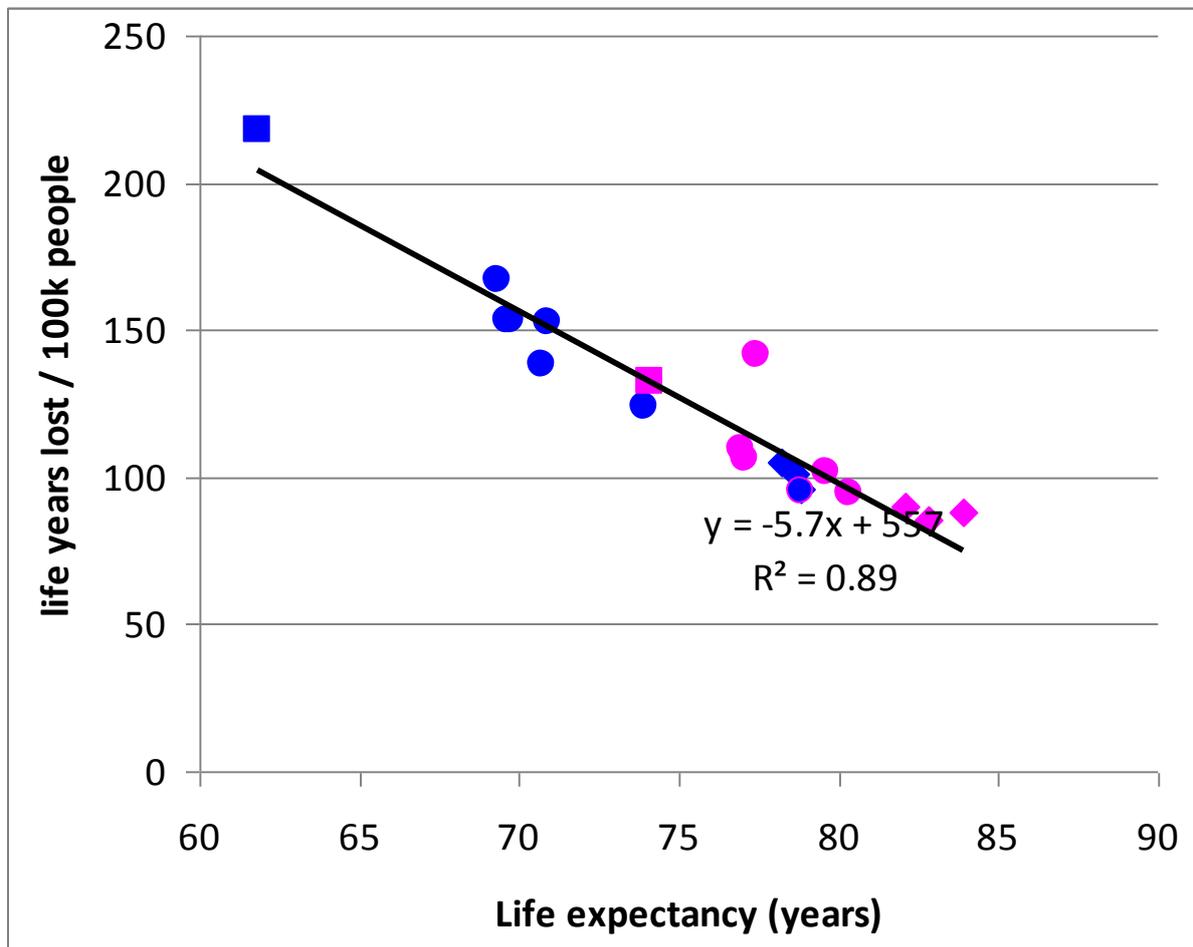


Figure 4. Relationship between life expectancy and life years lost per 100,000 people from a one-year change in exposure to PM_{2.5}. Key: Blue – male; pink – female; squares – Russian Federation; circles – Bulgaria, Czech Republic, Hungary, Poland, Slovakia, Romania; diamonds – England/Wales, Italy, Sweden.

It is noted that the adjustment for Russia, in particular is extremely high, and so its reliability may be questionable.

4.4. Acute mortality from ozone

The cohort studies preferred for assessment of PM related mortality did not show a clear effect of ozone at the time of the CAFE analysis, though Jerrett et al (2009) observed an effect of chronic exposure. However, prior to the conclusions of REVIHAAP and HRAPIE, it has been decided that it is appropriate to remain consistent with the existing analysis within GAINS, and the benefit assessment team's own long-standing practice. As a result, we include the acute mortality effect from short-term exposure to ozone from the time-series studies. Quantification within GAINS has again been guided by WHO/TFH, whose main conclusions were:

- Quantification should focus on ozone characterised as a daily maximum 8-hr mean, in relation to daily all-cause mortality.
- There is not robust evidence for the presence of a threshold for ozone effects, though quantification is less reliable at low ozone concentrations.
- With this caveat in mind, the main quantification of ozone effects on mortality should use the metric SOMO35 (sum of means over 35 parts per billion (ppb)). This measure represents accumulated exposure to concentrations greater than 35 ppb daily maximum 8-hr mean. This should not be taken as an indication that there are no effects under 35 ppb.
- A risk estimate of 0.3% increase in daily mortality (95% CI 0.1-0.4%) per $10 \mu\text{g}/\text{m}^3 \text{O}_3$ should be adopted.

The direct output from the analysis that follows these recommendations is the effect of ozone in terms of number of deaths brought forward, which can, in theory, be valued using the VSL approach. However, it is problematic to do so for two reasons. First, it implies that short term exposure to ozone is the primary cause of death: we would expect that those who die in such circumstances to have a pre-existing health condition that may not be related to pollution exposure at all. Second, it implies that individuals have no preference between dying a little early and dying many years early, which is self-evidently nonsense. On this basis we consider it appropriate to value ozone-mortality using the VOLY as it accounts for remaining life expectancy.

To quantify using the VOLY approach it is necessary to convert the number of deaths calculated using the acute exposure functions into the number of life years lost. This is difficult, as there is a lack of direct evidence for this part of the quantification. Some previous work (ExternE 1999a, Hurley et al. 2000) has involved 'conversion' of attributable deaths from time series studies to changes in life years using an estimate of 6 months per life. When originally made, this estimate was considered by many likely to be an overestimate, though attitudes have since changed. The peer review of the CAFE-CBA methodology thought that a larger value, on average, was warranted. A US evaluation of ozone and mortality has used an estimate of 12 months, and this was used for CAFE-CBA also. While there is no direct evidence to justify this figure, it seems an appropriate balance between those losing a short amount of life expectancy and the death of individuals who would have recovered and lived for significant periods. In CAFE it was suggested that other values could be investigated in sensitivity analyses, ranging from 2 to 18 months should ozone effects on health prove significant for any particular analysis. However, results showed that this was unnecessary as ozone effects were small relative to PM impacts.

WHO/TFH recognised that estimating effects only above a cut-off point is a very conservative approach to the estimation of the mortality effects of ozone. Consequently, it recommended a sensitivity analysis giving estimates with no cut-off point (or equivalently, with cut-point at

zero) as an upper bound on the true impacts. Again, this could be included in sensitivity analysis for the benefits assessment. The importance of this for a CBA will be limited, however, by the fact that key to the outcome of a CBA is the difference in impact between scenarios, rather than the total impact per se.

4.5. Infant mortality from PM

In Europe the effects of air pollution on mortality and other indices of infant health have been studied most extensively by Bobak and co-workers, initially in the Czech Republic, but also more widely (Bobak and Leon 1992, 1999; Bobak, 2000; Bobak et al., 2001). In quantifying the benefits to health of the US Clean Air Act, it has been recommended that infant mortality be included for quantification. This position is adopted here also.

As with the US analysis, the quantification is based on the cohort study by Woodruff et al (1997). The associations reported by Woodruff et al are with particulate matter, expressed as PM₁₀ (mean outdoor concentrations of PM₁₀ in the 1st two months of life) giving

Change in (all-cause) infant mortality of 4% per 10 µg/m³ PM₁₀

Results were based on a study of 4 million infants, where post-neonatal infant mortality was considered as death between the ages of one month and one year. This eliminates the first month of life when infants face the highest risk of death, but are also most likely to be receiving treatment that protects against exposure to air pollution. Results are described in terms of numbers of deaths, valued using an adjusted VSL (see below). Quantification using a VOLY approach is not considered appropriate given the anticipated reduction in life expectancy of deaths before the age of 1 year.

Results from the CAFE analysis suggested that the deaths of very few infants could be linked to air pollution compared to the deaths of older people. In part this was a function of very low infant mortality rates across the European domain considered for CAFE. However, it was noted that infant mortality rates in some countries (particularly Turkey) were much greater, so an apparently small increase in the size of the geographic domain considered could generate a major increase in the magnitude of this impact.

There is a wider issue of effects on mortality from acute exposure in children under five years; and, more generally, mortality effects (from chronic and acute exposure) in people under 30 years of age (note that the 'general' cohort studies are studies of adults, at ages 30 or more). This has not been addressed either in the CAFE-CBA work or in EC4MACS.

4.6. Valuation of mortality in adults

The valuation of air pollution related mortality has been the subject of much debate in recent years and so it is appropriate here to present a detailed review of the current literature. One topic of debate relates to the preference for using the Value of a Statistical Life (VSL) or the Value of a Life-Year (VOLY) and reflects the discussion above on the interpretation of the epidemiological evidence.

To date, the empirical basis for monetary valuation of the Value of a Statistical Life (VSL) has been much stronger than that of the valuation of a life-year-lost (VOLY). In the absence

of empirical studies, the VOLY was derived from the VSL, using methods that require assumptions about discounting the value of future life-years. However, the derivation of numbers of deaths (especially for chronic mortality analysis) is considered highly uncertain from the perspective of health impact assessment. The application of the full VSL to a short reduction in individual life expectancy is also questionable.

Hurley et al (2005) traced methodological developments in the valuation of mortality endpoints for the air pollution context in the period from the early 1990s to 2004, when the results from two new empirical studies were published. In work for the European Commission the results of the new work were considered alongside the conclusions of a workshop previously convened by the Commission (EC DG ENV, 2000). This section updates the discussion in that study by including evidence from a third new empirical study, recently undertaken as part of the EC DG Research New Energy Externalities Development for Sustainability (NEEDS) project (published as Desaignes et al, 2011). We place this evidence in the context of the two previous studies – sponsored by the European Commission and the UK's Department for Environment, Food and Rural Affairs (DEFRA).

Faced with a growing evidence base from three studies that utilise different methods and generate contrasting results, we need to undertake some sort of evaluation of the evidence in order to suggest appropriate values for use in air quality policy appraisal. Thus, the subsections below outline principal features of the methods applied in the alternative studies, and highlight key differences, before determining how the results might be weighted for use in EC4MACS.

4.6.1 The NewExt study

Method

As part of the European Commission-funded ExternE series of projects, the New Elements for the Assessment of External Costs from Energy Technologies (NEWEXT) project included a new empirical study of mortality risk valuation. This study, reported by Markandya and later published by Alberini set out to derive unit values to account in monetary terms for the incidence of premature death linked to air pollution in Europe. Values were derived from three surveys undertaken simultaneously in the UK, France and Italy, using a common survey instrument. The survey instrument adopted by the country teams in the UK, France and Italy had previously been developed using extensive face-to-face interviews in the USA, and was pre-tested in the USA, Japan and Canada before being used for full sample surveys in those countries.

The survey was computerised and self-administered. The survey protocol included people over the age of 40, with one third of the sample being over the age of 60. The three EU country teams each conducted a series of focus groups and/or one-to-one testing in order to adapt the instrument for the national contexts. Additionally, the French country team tested a series of variants to the questionnaire on samples of about 50 people.

The survey instrument was designed to elicit WTP for mortality risk reductions. Specifically, people were asked to value an immediate 5 in 1000 risk reduction, (the risk change being spread over the next ten years), an immediate 1 in 1000 risk reduction, and a reduction of 5 in 1000 to be experienced at age 70, in that order (Wave 1). (Wave 2 in the N. American studies reversed the order of the immediate risk changes). The France study also implemented the Wave 2 design, whereby the 1 in 1000 risk reduction was valued first. The size of these risk

changes are thought to be in the appropriate range for capturing the risk reductions associated with air pollution reductions.

Table 3. Sample size (N) and experiment design for the EU 3-country study

	UK	Italy	France
N	330	292	299
Study location	Bath	Venice, Genoa, Milan and Turin	Strasbourg
Experimental design	Wave 1	Wave 1	Wave 1 and wave 2

The North American team believed that there were compelling reasons for keeping the agent for the risk reduction and the payment vehicle completely “abstract”, as in this example. Whilst this departs from the NOAA panel recommendations, it was felt that there was sufficient evidence (see e.g. Hurd and McGarry, 1997 and Cropper et al, 1994) to show that respondents are willing and able to make choices among abstract life-saving programs allowing respondents to focus on the size of the risk reduction itself and the effect it has, thereby avoiding various potential biases linked to perception of specific risk agents. Moreover, making the risks specific to a context may result in reduced values since people may not believe that specific risks apply to them. In the specific case of reductions in air pollution, there are numerous non-health benefits, and benefits to others, which people may or may not factor into their valuation. It was argued that these factors may lead to distorted estimates of the value to the individual of the health benefits.

Results

Results for the 3-country pooled data are summarised below. For reference we also include the VSL estimates derived using the same survey instrument in Canada and US (Alberini et al., 2004). We focus on the WTP values for the 5 in 1000 risk reduction because previous testing in the North American context suggests that answers to the first question asked tend to be more reliable. It is also likely to be an easier size of risk change to effectively comprehend.

Rabl (2001) derives the changes in remaining life expectancy associated with the 5 in 1000 risk change over the next 10 years valued in this study, based on empirical life-tables. According to Rabl’s calculations, the extension in life expectancy ranges from 0.64 to 2.02 months, depending on the person’s age and gender, and averages 1.23 months (37 days) for our sample.⁶ On this basis, we can compute the 3-country WTP estimates of VSL to life year equivalents and thus derive the corresponding VOLY.

⁶ A change in the probability of surviving the next 10 years changes the probabilities of surviving all future periods, conditional on being alive today. The product of these future probabilities of surviving is a person’s remaining lifetime. Rabl’s calculations are based on an exponential hazard function, $h(t)=\alpha*\exp(\beta t)$, where t is current age, and α and β are equal to $5.09*E-5$ and 0.093 for European Union males, respectively, and $1.72E-5$ and 0.101 , respectively, for European Union females.

Table 4. NewExt Results for 3-country pooled data (€) and N. American applications of survey instrument. € price year 2005.

	3-country pooled estimates			Canada	US
	5:1000 risk change - immediate	1:1000 risk change - immediate	5:1000 risk change – latent	5:1000 risk change – immediate	5:1000 risk change - immediate
Value of Statistical Life (VSL)					
Median	1,087,845	820,533	38,309	526,746	728,700
Mean	2,238,150	3,077,891	103,684	971,253	1,603,140
Computed Value of One Life Year (VOLY)					
Median	58,088	37,436	17,478		
Mean	130,385	140,429	47,305		

Regressions on the 3-country pooled data show that income is significantly associated with WTP, a result that is consistent with expectations. WTP declines only for the oldest respondents in the sample, who hold WTP amounts that are approximately 25% lower than those of the other respondents, all else the same. However, the coefficient on the dummy for a respondent who is 70 or older is not significant at 5% and 1% levels; the French results suggest that such an age effect may be due to lower income during retirement. As in earlier studies, males have slightly lower WTP and so do people with higher levels of education. Persons who have been hospitalized for cardiovascular or respiratory illnesses over the last 5 years hold WTP amounts that are over twice as large as those of all others. The presence of cancer and chronic illnesses, however, does not influence WTP.

It should be noted that the North American studies had mixed results regarding the significance of age and health in determining WTP. Whilst the Canada study found that WTP falls by 25% for respondents over the age of 70 and was significant at 5%, the US study found a fall of 20%, which, however, was not statistically significant. The effect of having cancer in Canada was positive and significant, though no effect was identified in US. However, in both countries a history of chronic family illness meant a positive and statistically significant effect on WTP. In other words, if the WTP to reduce risk of premature death is not affected much by age, it would imply that the WTP for one life year would increase substantially towards old age. This effect is not captured in the VOLY computation from VSL.

4.6.2 The DEFRA study⁷

Method

The authors of this study conducted a contingent valuation survey in the UK to elicit willingness to pay for a reduction in air pollution that would bring four health benefits: (i) an extension in life expectancy for the respondent and everyone else in the respondent's household in normal health state (N), (ii) an extension in life expectancy that would benefit

⁷ Valuation of health benefits associated with reductions in air pollution. Final Report. May 2004. www.defra.gov.uk.

elderly people with heart or lung disease (P), (iii) a reduction in the number of admissions to the hospital that would be experienced by older people with lung disease, or younger people with asthma or other chest condition (H), and (iv) two or three days of breathing discomfort every year for persons with asthma or allergies or other chest conditions (D).

Regarding health endpoints (N) and (P), respondents were randomly allocated to one of three possible levels: one, three or six months. This allows the researchers to test for scope, i.e., to see if WTP increases with the length of the gain in life expectancy. The respondents were over the age of 18 and answered on behalf of their household. The sample size for each of the sub-samples was 165; the sample was taken from 41 postcode districts across the UK.

Respondents were told that air quality policies would result in higher prices of products and services, and were queried about their willingness to pay for the set of four reductions using payment cards. The amounts of money people were queried about were annual payments that they would have to make, every year, for the rest of their lives. The respondent was then asked to break down WTP into the amounts that would be allocated to each of the four health endpoints.

The survey questionnaire was administered in person to the respondent using a computer-assisted protocol.

Results

The results from the DEFRA study are presented in Table below. Arguments have been put forward in the study for emphasis on the results for 1 month. One argument is that “on the basis of current epidemiological knowledge, the kinds of measures which are realistically available to policymakers are more likely to generate benefits of the 1-month-per-person kind than of a greater magnitude, so that the figure from the 1 month sub-sample is the most relevant” (DEFRA, 2004). We show the 1-month results, together with those for 3 months and 6 months. Indicative VSL-equivalents are also presented, calculated by assuming 40 years of remaining life expectancy.

Table 5. Values per month (mean) (€, price year 2003)

	1 Month Normal	3 Month Normal	6 Month Normal	1 Month Poor
Annual WTP per person for one year	538			149
Value of one life year	41,994	14,332	9,180	11,581
Value of Statistical Life	1,680,000	573,300	367,180	463,245

4.6.3 The NEEDS Study⁸

Method

Within this study, Desaignes et al (2011) undertook a contingent valuation survey to value life expectancy changes associated with air pollution in a number of different EU countries. The survey was administered in a major city in each of nine different countries - UK, France, Poland, Czech Republic, Hungary, Switzerland, Spain, Germany and Denmark - with a view

⁸ To be found at: http://www.needs-project.org/docs/results/RS1b/NEEDS_RS1b_D6.7.pdf

to pooling the data to estimate a common VOLY from air pollution reduction for use across the EU. As such, the study focuses on gains in life expectancy in normal health (chronic effects). The total number of completed interviews was 1,463.

Samples	Observations
Switzerland	179
Czech Republic	229
Denmark	136
Spain	100
France	101
Hungary	118
Poland	150
UK	150
Germany	300
Pooled data	1,463

The first stage in the questionnaire asked respondents to consider the effect of pollution on their health and whether it was of general concern to them. The aim of these early questions was to provide initial context for the subsequent valuation questions and to allow respondents to begin to consider explicitly the effects of air pollution on health. Subsequent questions developed this aspect further, concentrating on their knowledge of the effects of air pollution on health, air pollution-related illnesses in their family, their general health and lifestyle, specifically exercise and smoking.

In the second stage of the questionnaire the respondent was given information on average life expectancy in the various countries and how air pollution affects life expectancy. Attention was also drawn to their own life expectancy, given how old they were at the time of interview and factors which affect individual life expectancy such as genetic, behavioural and environmental conditions. Two potential policies that could reduce air pollution and hence generate gains in life expectancy of 3 months and 6 months respectively were outlined in the third stage of the questionnaire.

The remaining information focused on presentation of a schematic graph used in the questionnaire to illustrate the notion of a survival curve whereby a person's chance of surviving to the next year, conditional on them reaching the current age, falls as we age until at some point, i.e. death, that person has no chance of further survival. Thus, each person has a survival curve associated with current levels of air pollution. Were the level of air pollution to change, the survival curve would shift up or down depending on whether air pollution is reduced or increased, respectively. The respondents used randomly picked payment cards to generate WTP estimates for the life expectancy increases.

Results

The WTP results for the LE changes showed a ratio of 1.29 between WTP for 6 months LE increase and 3 months LE increase.

Sample	3 Months WTP		6 Months WTP	
	Protesters deleted	Protesters and outliers deleted	Protesters deleted	Protesters and outliers deleted
EU 16	30,318	25,762	47,513	40,133
New Member Countries	19,339	19,339	29,228	29,228
Pooled – EU25	26,258	23,376	40,726	36,071

In order to account for differences in population sizes among the 9 countries in the sample, the pooled countries VOLY was re-estimated, weighting it by the populations in the NMCs and the EU16 (EU15+Switzerland) and by an average of the VOLYs derived by assuming:

- i) remaining LE of the respondents' average age, and
- ii) varying LE of each individual respondent.

The central resulting VOLY estimates are derived from the 3 month WTP results, on the basis that the individuals' budget constraint will constrain the WTP for the 6 month LE increase, and because air pollution policies are more likely to cause an improvement in life expectancy of 3 months rather than 6 months. Results are rounded to:

EU 16:	41,000 euro
New Member Countries:	33,000 euro
EU25 (EU26, including Switzerland):	40.000 euro

For completeness, the resulting VOLY estimates derived from the 6 month WTP results are rounded to:

EU 16:	27,000 euro
New Member Countries:	22,000 euro
EU25 (EU26, including Switzerland):	26,000 euro

Concern with the health effects of air pollution is statistically significant at 1% for the 6 month LE gain and 10% for the 1 month LE gain. Income was positive. Age is positive but not significant at 10%, at least. Just under 24% of the sample had zero bids whilst 9.6% of the sample were judged to be protest bids.

4.6.4 OECD (2012)

It has been noted that there is a significant difference between the VSL adopted for European policy work and that adopted in North America, with the latter being significantly higher as a rule. The review recommended the adoption of estimates based on the willingness to pay literature rather than the wage-risk studies that appear to have been preferred in the USA, and which are the source of the higher valuations. The OECD performed a meta-analysis from which a best estimate of \$3.6 million (€2.8 million) was recommended.

4.6.5 Evaluation of the new mortality valuation studies

The study outlines presented above highlight that there are a number of methodological issues that are far from being resolved at a generic level. The most prominent amongst these are as follows:

Respondents' understanding of the "good" being valued

The survey instruments in the three studies described above have been subject to repeated criticism that respondents do not understand what they are being asked to value, despite the best efforts of the teams undertaking the surveys. For example, it is generally considered that whilst people can declare fairly realistic WTP values for goods that they know or that they can understand, the risk of death is something they are not at all accustomed to thinking about in monetary terms. Furthermore, it is recognised that people have a great deal of trouble comprehending and valuing small risks. The same arguments are made against respondents' ability to value changes in life expectancy over the whole remaining lifetime (i.e. not simply valuing an extra e.g. 6 months at the end of a lifetime) and, in the case of latent risks, in future time periods. Susceptibility to this problem is illustrated in all three studies that show no proportionality between different sized changes and so fail the internal scope test.

The treatment of (air quality) context in the survey design.

A methodological issue that has dominated discussion in recent survey designs is the question of whether it is better to include the air quality context in the survey instrument but risk free-riding and altruism effects on WTP, (as in the NEEDS survey) or keep the survey context-free but run the risk of excluding context-specific elements of the WTP (as in the NEWEXT survey). Ready *et al.*, (2004a) argue that studies that have used stated preference methods to evaluate health episodes tend to be vague in the survey instrument about the cause of the illness that causes the endpoints; how the illness would be avoided or how the improvement would be paid for. The resulting WTP values are then assumed to be applicable for policy analysis of any programme that results in changes in numbers of that type of episode. However, the NOAA⁹ expert panel on contingent valuation concluded that respondents could not reliably answer contingent valuation questions about environmental goods unless the hypothetical programme to provide the good is described in detail. Regarding health-related impacts of environmental programmes, the cause of the ill health and the way it would be treated are inherent to the environmental programme.

The risk involved in evaluating the illness endpoints in the context where the risks are generated is that the focus of respondents can be deviated from the endpoints themselves to the cause of the endpoints. For example, it has been found that a major reason for respondents refusing to engage in an air quality programme that would prevent them suffering some health symptoms was that respondents did not think they were polluters and should not suffer the financial consequences.

WTP elicitation format

It remains to be seen whether there is robust evidence of starting point bias being introduced by the use of dichotomous choice in the survey instrument, compared with loss of incentive compatibility. The WTP elicitation format used in the NEEDS study is the payment ladder, which allows for a range of uncertainty over the value respondents place on the commodity being valued. Its advantage to the more familiar elicitation format used in the NEWEXT study – dichotomous choice format with or without follow-up questions, recommended by the

⁹ U.S. National Oceanic and Atmospheric Administration.

NOAA expert panel on contingent valuation – is that respondents may know with certainty the values they would pay and those that they wouldn't, but still have values that they are not sure about. In addition, when respondents are faced with a higher range of values to say 'yes' or 'no' to for that payment, there is the possibility that those respondents give more consistent answers since they spent more time thinking about the decision.

Payment vehicle

The payment vehicle used in the NEWEXT study is a medical product or action that tries to ensure that respondents think about their own risks (rather than any one else in the household since the medicine/treatment is individual) and so make the valuation exercise similar to the more familiar exercise of buying goods in a supermarket or pharmacy. However, this approach departs from the real context of air pollution and may introduce other biases. On the other hand, as noted above, the air pollution context adopted in the Defra and NEEDS studies gives rise to the potential for free riding, lumping a range of health and other benefits together and altruistic preferences that may lead to double-counting.

4.6.6 Summary of mortality values proposed for EC4 MACS

The logic adopted in the CAFE programme was as follows. It was decided to use the NEWEXT study results as the basis for mortality valuation, on the basis that, unlike previous work, the studies were designed specifically to measure the WTP for mortality changes associated with air pollution. The results of the Defra study were excluded on the basis that the results were derived from one EU country only, thereby reducing its applicability in representing the EU population. Additionally, the methodology developed in the Defra study has to some extent been incorporated into the NEEDS study that samples over nine countries. The OECD work on the VSL certainly makes a major contribution to the debate, given that it represents a much larger body of work.

The NEWEXT sampled in three EU Member States and the total size for the 5:1000 immediate risk change question was 930. It values a risk change and therefore allows a VSL to be computed directly. As shown above, however, WTP estimates for the risk changes need to be converted to life year equivalents in order to derive a corresponding VOLY. By contrast, the NEEDS samples are taken from nine countries to give a total pooled sample size of 1463. The study also directly values a change in life expectancy. But the evaluation discussion above shows that both studies are susceptible to charges of methodological weakness. As a consequence, it is very difficult to argue for the merits of one set of results over the other set. It is notable that the standard errors around both sets of estimate show that each set encompasses the other. It is suggested therefore that the values adopted in the CAFE analysis need not necessarily be replaced by those generated by NEEDS. Further to this there is a formal process for revision of these numbers for policy use, recognising that they are broadly applicable to many policy fields. Accordingly, it was decided to continue using the NEWEXT values (median and mean VOLY, median and mean VSL) as the basis for mortality valuation. However, sensitivity analysis using the €40k VOLY from NEEDS/Desaigues et al (2011) and the €2.8 million VSL from OECD (2012) is also recommended to test effects on the analysis and conclusions reached.

A number of issues deserve mention at this point:

- It is appropriate to use the VOLY, as this is the preferred approach from the health impact assessment analysis. For the present, however, because of the limited evidence base for

the VOLY approach we also quantify separately using the VSL, which has a more established empirical basis.

- It is recommended to use the results from the NEEDS and NEWEXT studies which relate to WTP in normal health since the section on epidemiology above suggests that this is most appropriate to describing the chronic impact.
- We also need to consider adjustment for the quality of the life lost. The NewExt study finds that the fact that a respondent has a chronic heart or lung condition does not influence WTP *per se*. However, those persons who have been hospitalized for cardiovascular or respiratory illnesses over the last 5 years have WTP amounts that are, everything else being the same, roughly twice as large as those of all others. Therefore, as a sensitivity it may be appropriate to apply a multiplier of two, or at least to note that the omission of such a factor could bias results downwards.
- The question of whether to use the VSL or VOLY approach for valuation is more problematic for dealing with the acute impacts of ozone. For these acute mortality effects of ozone (from the time-series studies), the health impact can best be characterised as “deaths brought forward” (generally by a short time compared to deaths from transport accidents). The time series studies provide results in terms of changes in the number of daily deaths associated with air pollution. Aggregated, these results can be represented as the number of deaths per annum whose immediate life shortening was attributable to air pollution in the days immediately preceding death. In at least some of these cases, the actual loss of life is likely to be small, though there is no direct evidence on the average loss of life. From the perspective that the period of life lost to this ozone effect is small and will occur most commonly in people with existing health problems (i.e. a lower quality of life), it is the view of the study team, that for such cases, the use of a full Value of Statistical Life is inappropriate and we strongly argue in favour of the VOLY approach. There is a caveat, however, relating to the work by Jerrett et al (2009) suggesting a chronic effect of ozone exposure, the omission of which at the present time may be unduly conservative.
- The WTP estimates are not found to be age-dependent in either study and so we do not adjust for age.
- The use of multiple values for a single endpoint is problematic for the presentation of analysis. Ideally, we would use a single value supported by a range that could be carried through the uncertainty analysis.

There are some further issues, the principles of which apply to the valuation of other cost and benefit elements to be included in the analysis.

- First, comparison of the WTP results for the immediate risk reductions and the latent risk reduction in the NewExt allows calculation of implicit discount rates of 5%, 6% and 10% for Italy, France and the UK respectively. A pure efficiency-based approach would suggest adoption of these rates for the different countries – just as an efficiency-based approach would suggest using the different WTP values for the different countries. For consistency with the discounting of other costs and benefits we prefer to adopt the common rates used in Commission’s impact assessment of 4%. Alternative rates could be used in sensitivity analysis, though there is probably little demand for it.
- Whilst the efficiency criterion alone suggests using different WTP values for different Member States, this has not been carried out by any analysis at European level. This is mainly because it would not be politically appropriate to use different values and also because there is the practical challenge of getting such values from Member States.
- It is noted that incomes are expected to rise over time, so should the WTP valuations be expected to increase. At issue here is whether the two increase at the same rate. A

number of studies have reported a positive relationship between income and WTP values, including the NewExt study, the NEEDS study, Hammitt and Liu (2004) etc. Whilst we recognise this approach to be correct in itself, we are again constrained by the informational requirement that would arise were we to apply this mechanism across all costs and benefits (with potentially differing income elasticities). We will therefore adopt constant unit values across time. At the time of adoption this was considered likely to result in some underestimation of benefits, as it was thought likely that WTP would increase when disposable income increases. However, in the current economic climate such an assumption may be incorrect.

- Related to this, there is also an issue of discounting. Some of the benefits in life years will occur in the future (potentially over a much longer time frame than for the CAFE scenario), so it is important to consider the discounting of benefits, using a discount rate that is consistent with the other parts of the CAFE analysis (e.g. on costs) and, indeed, analysis for the European Commission more generally. This is a social rate of time preference of 4%¹⁰. For longer-term analysis (e.g. working with life table over a 75 – 100 year times-scale) a different approach is often taken in environmental cost-benefit analysis. This uses a pure rate of time preference (PRTP), for example of 1.5%¹¹. The logic for this is that benefits, such as a change in the risk of death, might be seen as having a broadly constant utility value over time, regardless of changes in income. If so, then such future benefits can be valued in current values and discounted at [the pure time preference rate], so avoiding the need to calculate separately a rate of increase in their value over time. However, we have not discounted future life years lost (or gained), though the assumption of no lag phase reduces the importance of any discounting significantly. It will be considered in future revisions of the methodology. The lack of discounting provides some bias to an over estimation of the benefits.
- Taking together the fact that WTP is likely to increase over time (either with the same rate as GDP or somewhat higher or lower) and the fact that future benefits are not discounted, an implicit – and most likely realistic – assumption in the methodology is that these two effects cancel each other out.

4.7. Valuation of Mortality in Infants and Young Children

There is a paucity of research in this area, and the conflicting results from the limited existing literature leaves little guidance for policy makers on how to value health risks to children.

For example, due to the lack of empirical research on VSL, most economic analyses rely on adult VSL for children's health effects. Indeed, in the United States EPA's 'Children's Health Valuation Handbook' recommends that "with few child health valuation studies available, analysts may need to rely on transferring adult benefits to children until more information becomes available" (US EPA 2003, pp. 1-6). In the following paragraphs we review the evidence that can be considered and make some suggestions as to what values can be used in the first instance.

¹⁰ Related to the paragraph above, when considering a much longer time-frame, the effects of incomes rises over time, and WTP rises, are also relevant.

¹¹ The value of a 1.5% PRTP is broadly consistent with the current EC recommended discount rate of 4% social rate of time preference (assuming average GDP per capita growth of 2.5%).

4.7.1 Methodological issues

A number of methodological and empirical issues must be resolved. The first methodological issue is which perspective to adopt. Generally, economists prefer to derive WTP values for health risk reductions from the willingness of individuals to pay for risk reductions that affect themselves. This approach would clearly present difficulties in the case of children, as children have neither the maturity nor the financial resources to clearly define their willingness-to-pay. In short, the basic tenets of welfare economics cannot reasonably be assumed to represent children. Therefore, an alternative perspective has to be adopted from which to estimate child health values.

There are three potential perspectives from which preferences for children's health risks might be elicited:

- (i) that of society (parents and non-parents),
- (ii) that of adults placing themselves in the position of children, and
- (iii) that of parents assessing risks faced by their own children (Dockins et al, 2002).

Obtaining a societal WTP presents problems of double-counting due to altruism (Jones-Lee 1991; Jones-Lee 1992). The parental perspective has the advantage that literature is available, albeit sparse. We report on this literature below.

Other methodological issues making transfer of values from adults to children difficult are:

- Differences in the nature and/or extent of impact risks. There is, for example, considerable uncertainty in the epidemiological literature as to the likelihood and magnitude of health impacts on children (Tamburlini, 2003).
- The potential importance of altruism. This is clearly important when parents assess their children's health risks though the identification of the type of altruism involved in determining WTP is difficult.
- The relationship between WTP and age. This would be important for example were a measure of life years to be used in the valuation – particularly when the child's life expectancy is uncertain.
- Household composition and structure. Within the parental perspective approach it is likely that the nature of the household formation will influence WTP. For example, WTP has been found in some studies to be less for children in larger families, and for single parent families to be WTP more for avoiding illness in their child than dual parent families.
- Discounting. Subjective discount rates may be different for adults and their own WTP compared to those for their children.

4.7.2 Empirical issues

A number of studies have examined possible differences of values between adults and children, but their findings have been mixed. The majority of studies find that the value of children's health benefits is higher than those of adults (Lewis and Charney, 1989; Cropper et al, 1994; Liu et al, 2000; Dickie and Ulery, 2001). However, other research has generated estimates of WTP for child and adult health that are similar (Blomquist, 2003; Mount et al, 2001) and one study estimates the value of statistical life for a child that is lower than the value of a statistical adults life (Jenkins et al, 2001). The findings of these studies that are of most relevance in the context of valuing the avoidance of children's premature death are presented in the table below.

It should be emphasized that none of the above listed studies consider the EU context and that there are many questions and obstacles surrounding the validity of transferring mortality risk values across countries. Nevertheless, some conclusions can be drawn:

- Parents are more willing to pay to reduce their children’s health risks than their own. The estimated marginal rate of substitution (MRS) is generally greater than one, and is typically about 2.
- Values for younger children are generally found to be higher than for older children.
- Values for the reduction of children’s mortality risk are greater than the values for the reduction of morbidity risk. This is consistent with the WTP values for adults and is consistent with economic theory.

The methodological difficulties outlined above that remain unresolved – combined with the differences between studies and the spatial and temporal transfer issues - make it very difficult to transfer these results to our current context with any great confidence.

Table 6. Valuation of child mortality

Author/Year	Valuation method	Good/service to be valued	Value (€, 2000)	Country
		Premature Mortality	VSL	
Joyce (1989)	COI	Pre-natal care Neonatal care (age <1)	52,000-175,000 85,000-4,100,000	US
Carlin & Sandy (1991)	Revealed preference	Expenditure on car safety (age <5)	880,000	US
Blomquist, Miller and Levy (1996)	Revealed preference	Expenditure on car safety (age <5) Adult	4,100,000-6,600,000 3,100,000-5,100,000	US
Mount, Weng, Schulze and Chestnut (2001)	Revealed preference	Expenditure on car safety (child) Adult (age 40) Adult (age 70+)	8,100,000 7,900,000 5,700,000	US
Jenkins, Owens and Wiggins (2001)	Revealed preference	Expenditure on bicycle safety (child age 5-9) (child age 10-14) Adult	3,200,000 3,100,000 4,700,000	US
		Morbidity		
Liu et al. (2000)	Stated Preference	Influenza Child Adult	63 41	Taiwan
Agee and Crocker (2001)	Revealed preference	10% increase in health status Child Adult	500 275	US
Dickie & Ulery (2001)	Stated Preference	Acute illness (one symptom): Child Adult Acute bronchitis Child Adult	165-385 110-180 440 220	US
Dickie & Gerling (2001)	Stated Preference	1% reduction in prob. of non-melanoma skin cancer Child Adult	3.5 1.4	US

Derived from Scapecchi (2003)

Accordingly, it is recommended to apply a multiplier of 1.5 to the adult VSL estimates (range 1 to 2) for application to children. Members of the consultancy consortium are involved in an EC-funded project that has the remit to undertake new empirical work on valuation of children's mortality risks in Europe. However, the results of this study will not be available to inform this phase of the CAFE research programme. The text above makes it clear that there is quite some uncertainty in these estimates. However, they make little difference to the final results, given the small number of mortality cases involved.

4.8. Morbidity from PM and ozone

4.8.1 Approaches to quantification

The basic approach to estimating the effects of air pollution on human morbidity is similar to what is done for 'acute' mortality; i.e. use a concentration-response (C-R) function expressed as:

- i. % change in frequency of occurrence of an endpoint (relative risk (RR) of new or 'extra' cases) per $10 \mu\text{g}/\text{m}^3$ PM₁₀ or ozone; link this with:
- ii. the background rates of the endpoint (new cases per year per unit population – say, per 100,000 people) in the target population
- iii. the population size
- iv. the relevant pollution increment, expressed in $\mu\text{g}/\text{m}^3$ PM₁₀ or ozone

and express the result as estimated new or 'extra' cases per year.

Note that where neither the C-R function nor the background rates vary spatially, then these can be combined into a single *impact function* expressed as:

$$\begin{array}{c} \text{number of (new) cases per unit population (say, per 100,000 people)} \\ \text{per } 10 \mu\text{g}/\text{m}^3 \text{ PM}_{10} \text{ or ozone per annum} \end{array}$$

This impact function can then be linked, as before, with population size and the relevant pollution increment to give estimated impacts.

Where data on background rates of morbidity are unavailable, or in practice too difficult (too resource-intensive) to collate, one approach is not to attempt quantification. However, failure to quantify generates a systematic bias to under-estimation of health effects.

Alternatively it is generally possible to:

- estimate an impact function from where the relevant epidemiological studies were carried out; and
- transfer and use that impact function for quantification in the target population.

This alternative approach allows some quantification, albeit one that is less reliable than if suitable background data were readily available. It is, however, well-established in health impact assessment practice.

The strengths and weaknesses of these approaches and other information relating to the quantification of morbidity effects are discussed in volume 2 of the original CAFE-CBA methodology (Hurley et al, 2005).

4.9. Morbidity impacts to be quantified

Quantification in CAFE was performed against a variety of health endpoints, as shown below. The NEEDS study (Torfs et al, 2007) reviewed the literature and came to similar conclusions as had been reached in CAFE earlier. Below are listed the functions used for quantification of morbidity impacts, more complete details including full references and the rationale for selection of specific functions in preference to others are given by Hurley et al (2005) and Torfs et al (2007).

Reference is made to ‘core’ and ‘sensitivity’ functions, the latter giving either higher damages than core, or bringing in additional endpoints. These sensitivity functions were little used in CAFE, though are retained here. They add only a small amount to total damage estimates.

Whilst it is well-established that ambient air pollution, especially ambient PM, adversely affects lung function and measures of cardiovascular functioning, neither effect is quantified here as results are not amenable to monetary valuation. Their omission may, however, give an incomplete picture of the adverse effects of air pollution on health. The existence of these effects does, however, add plausibility to the findings linking air pollution and more serious cardiovascular outcomes (mortality, hospital admissions) that are proposed for quantification.

4.9.1 Chronic exposure

1. New incidence of chronic bronchitis

The only impact quantified in relation to chronic exposure to air pollutants in CAFE was new incidence of chronic bronchitis. Two functions were recommended, one for the ‘core’ analysis and a second for ‘sensitivity’ analysis

- Core: 26.5 (95%CI -1.9, 54.1) new cases of chronic bronchitis per 10 $\mu\text{g}/\text{m}^3$ PM_{10} per 100,000 adults
- Sensitivity: 53.3 (95%CI -1.7, 113.4) new cases per 10 $\mu\text{g}/\text{m}^3$ $\text{PM}_{2.5}$ per 100,000 adults

The relevant population-at-risk is the population of all adults, aged 27 years or more, who do not have chronic bronchitis.

4.9.2 Acute exposure

A considerably larger range of effects associated with acute exposures is proposed:

2. Respiratory hospital admissions (RHAs):

- Core: 7.03 (95% CI 3.83, 10.30) emergency RHAs per 10 $\mu\text{g}/\text{m}^3$ PM_{10} per 100,000 people (all ages)
- Core: 12.5 (95% CI -5.0, 30.0) emergency RHAs per 10 $\mu\text{g}/\text{m}^3$ O_3 (8-hr daily average) per 100,000 people at age 65+

An estimate can also be made for RHAs attributable to ozone exposure for the age range 15-64. However, the observed risk is small, the observed relationship was not statistically

significant, and representative background rates were not easily available. Inclusion of this effect would make a negligible difference to overall results.

3. Cardiac hospital admissions (CHAs)

- Core: 4.34 (95% CI 2.17, 6.51) emergency cardiac hospital admissions per 10 $\mu\text{g}/\text{m}^3$ PM_{10} per 100,000 people (all ages)

The increased mortality associated with daily variations in ozone includes excess mortality from cardiovascular causes. There is some evidence, e.g. from studies in London, that daily ozone is associated with increased cardiovascular admission. However, the evidence overall does not strongly support such a relationship and so no attempt is made to quantify this pathway.

4. Consultations with primary care physicians (general practitioners) in adults

- Sensitivity: 1.18 consultations (95% CI 0, 2.45) for asthma, per 1000 children aged 0-14 per 10 $\mu\text{g}/\text{m}^3$ PM_{10}
- Sensitivity: 0.51 consultations (95% CI 0.2, 0.82) for asthma, per 1000 adults aged 15-64 per 10 $\mu\text{g}/\text{m}^3$ PM_{10}
- Sensitivity: 0.95 consultations (95% CI 0.32, 1.69) for asthma, per 1000 adults aged 65 and over per 10 $\mu\text{g}/\text{m}^3$ PM_{10}
- Sensitivity: 4.0 consultations (95% CI -0.6%, 8.0%) for Upper Respiratory Symptoms (URS) (excluding allergic rhinitis), per 1000 children aged 0-14, per 10 $\mu\text{g}/\text{m}^3$ PM_{10}
- Sensitivity: 3.2 consultations (95% CI 1.6, 5.0) for URS (excluding allergic rhinitis), per 1000 adults aged 15-64, per 10 $\mu\text{g}/\text{m}^3$ PM_{10}
- Sensitivity: 4.7 consultations (95% CI 2.4, 7.1) for URS (excluding allergic rhinitis), per 1000 elderly people, aged 65+, per 10 $\mu\text{g}/\text{m}^3$ PM_{10}

Further functions were identified to link ozone exposure to consultation rates, though were not considered reliable enough for inclusion in the analysis.

5. Restricted activity days (RADs) and PM

- Core: 902 (95% CI 792, 1013) RADs per 10 $\mu\text{g}/\text{m}^3$ $\text{PM}_{2.5}$ per 1,000 adults per year at age 15-64
- Sensitivity: As core, but applied to all ages

For valuation it is useful to distinguish RADs by severity. It is assumed that two thirds are minor RADs. Work Loss Days (WLDs) are calculated as:

- 207 WLDs (95% CI 176-238) per 10 $\mu\text{g}/\text{m}^3$ $\text{PM}_{2.5}$ per 1000 people aged 15-64 in the general population

6. Minor restricted activity days (mRADs) and ozone

- Core: 115 (95% CI 44, 186) per 10 $\mu\text{g}/\text{m}^3$ ozone (8-hr daily average) per 1000 adults aged 18-64 per year
- Sensitivity: As core, but applied to all ages

7. Use of respiratory medication by people with respiratory diseases

- Core: 180 (95% CI -690, 1060) days of bronchodilator usage per 10 $\mu\text{g}/\text{m}^3$ PM_{10} per 1000 children aged 5-14 years meeting the PEACE study criteria (approximately 15%)

of children in Northern and Eastern Europe and 25% in Western Europe meet the inclusion criteria)

A function was identified in CAFE to link ozone with an increase in days of bronchodilator usage amongst children. The function was not used because of concerns over reliability, though it was stressed that medication use is an important health-related endpoint, so further research in this area is highly desirable.

- Core: 912 (95% CI -912, 2774) days of bronchodilator usage per year per 10 $\mu\text{g}/\text{m}^3$ PM_{10} per 1000 adults aged 20+ with well-established asthma (say, 4.5% of the adult population)
- Core: 730 days (95% CI -255, 1570) per 10 $\mu\text{g}/\text{m}^3$ O_3 per 1000 adults aged 20+ with persistent asthma (say, 4.5% of the adult population)

8. Symptom days

- Core: 1.86 (95% CI 0.92, 2.77) extra symptoms days per year per child aged 5-14, per 10 $\mu\text{g}/\text{m}^3$ PM_{10}
- Core: 1.30 (95% CI 0.15, 2.43) symptom days i.e. lower respiratory symptoms, including cough, per adult with chronic respiratory symptoms, per 10 $\mu\text{g}/\text{m}^3$ PM_{10} where we estimate that 30% of the adult population qualify as having chronic respiratory symptoms
- Core: 343 (95% CI 6, 692) symptom days per 1000 people at risk (all ages) per 10 $\mu\text{g}/\text{m}^3$ O_3 (8-hr daily max)

4.10. Valuation of morbidity

In reviewing the morbidity health end-points we draw particularly on the ExternE work and Ready et al. (2004).

To clarify what we are valuing: the starting point for the valuation of health end-points is the identification of the components that comprise changes in welfare. These components should be summed to give the total welfare change, assuming no overlap between categories. The three components include:

- Resource costs* i.e. medical costs paid by the health service in a given country or covered by insurance, and any other personal out-of-pocket expenses made by the individual (or family).
- Opportunity costs* i.e. the cost in terms of lost productivity (work time loss (or performing at less than full capacity)) and the opportunity cost of leisure (leisure time loss) including non-paid work.
- Dis-utility* i.e. other social and economic costs including any restrictions on or reduced enjoyment of desired leisure activities, discomfort or inconvenience (pain or suffering), anxiety about the future, and concern and inconvenience to family members and others.

The welfare changes represented by components (i) and (ii) can be proxied using market prices that exist for these items. This measure - in best practice - needs to be added to a measure of the affected individual's loss of utility, reflected in a valuation of the willingness-to-pay/accept (WTP/WTA), to avoid/compensate for the loss of welfare associated with the illness.

Note that there is the possibility of overlap between components since, for example, the individual will include both financial and non-financial concerns in his/her assessment of loss of welfare. Financial costs are often not borne fully by the individual but are shared through health insurance and public health care provision. Thus, we assume here that the financial costs are separable and measured in component (i). If this is not the case, then a part of the dis-utility measured in the WTP estimate will be incorporated in the private medical costs associated with treatment (or prevention) of the health end-point, and the total valuation should be reduced by an equivalent amount.

4.10.1 Health care resource costs

The generic unit costs for hospital-based health care are presented in the table below. The data has been derived from Netten and Curtis (2000), and MEDTAP International, reported in Ready et al. (2004). Since this data is based on public health care provision it is exempted from indirect taxes and is therefore expressed at factor cost. It has not been possible to derive unit cost data for all EU countries, but mean values calculated from the available data are presented and can be used as a first proxy for EU countries that currently do not report such values. Generic hospital costs are the average costs of a wide variety of specialist treatments, for use when precise information about the nature of the individual's hospital contact is not known.

The out-patient value for the UK is significantly higher than those in the other countries listed. This suggests that a different cost definition may have been used in its derivation – though this has not yet been established. The mean value, excluding the UK value, is €25, compared to the value of €36, when the UK figure is included. We suggest, for the present, that the higher value should be used as the central value, with the lower figure used for sensitivity analysis.

For cardiology, the inpatient unit cost is 1.92 times higher than the generic unit cost. This multiplier may then be applied when heart-related conditions are considered, in the discussion of end-points below.

Table 7. Generic Unit Hospital Health Care Costs (€ 2000 prices)

Country	Emergency Room/ Out-patient: cost/visit	Hospitalisation: cost/ inpatient day
Belgium	21	266
France	32	414
Germany	26	354
Italy	22	283
Netherlands	33	431
Spain	30	381
UK	106	364
Mean (EU)	39	357

Source: Netten and Curtis (2000), Ready et al. (2004)

Other unit cost data for more minor health conditions are presented in the discussion of the individual health end-points below.

4.10.2 The costs of absenteeism

CAFE used information on the costs of absenteeism contained in a report by the Confederation of British Industry (CBI, 1998). This report is the outcome of a survey on absence conducted by the CBI. Recommended values across the EU were €64, 97 and 288 per day of absence for low, central and high values respectively. The upper bound contained indirect as well as direct costs, the former being rather uncertain. Under EC4MACS a revised value has been adopted of €130/day based on an update to the original CBI report (CBI, 2011), in which it reports a median total cost for each absent employee of £760 per year and average rate of absence of 6.5 days per year. This value appears to include indirect as well as direct costs, but at a much more modest level than the 1998 survey.

4.10.3 Hospital admissions

Respiratory hospital admissions are one of the most widely-studied health endpoints, in Europe and internationally. Their quantification raises important questions about pollution mixtures and background rates of hospital usage. Results from other studies suggest however that the monetary value of their impacts is not high, compared with mortality from long-term exposure.

Ready et al. (2004) have estimated a WTP for respiratory hospital admissions in a survey-based approach (the contingent valuation method) where the patient stays in the hospital receiving treatment for three days, followed by five days at home in bed. Combining WTP with productivity losses and costs of hospitalisation for three days gives a total economic estimate of €2,220 per Hospital Admission from respiratory distress. This estimate is very similar to that derived by Otterström et al (1998) for a general HA episode (i.e. HAs independent of whether this is for a respiratory, congestive heart failure, ischaemic heart disease or cerebrovascular HA. We therefore adopt this common value for these end-points.

4.10.4 GP (General Practitioner) visits: Asthma & lower respiratory symptoms

Ready et al. (2004) found a WTP to avoid a day of asthma attack (excluding medical care and lost productivity costs) of €70, €145 and €307 per day for adult non-asthmatics, adult asthmatics and asthma attack among the respondents' own children, respectively. These were the values for a sample of respondents that were asked to express their WTP to avoid one additional day of asthma attack (in addition to what they had experienced the last 12 months). The corresponding asthma daily values for a sample that were asked to value an additional day to 14 days were €15, €16 and €44 respectively. The study suggests using the marginal day value of €16 as a central unit value.

Netten and Curtis (2000) give unit values for the resource costs of the GP in the UK. Here, we use these as representative of typical EU costs. These vary between €26 and €44 depending on whether the consultation period is 9.36 minutes or 12.6 minutes - the two unit periods suggested - and whether qualification costs are included. We assume that the longer period is more realistic for this condition. A value of €44 should therefore be added to the WTP values identified in the previous paragraph.

For lower respiratory symptoms a value of €40 may be used. This value was derived for the symptom described as "a persistent phlegmy cough occurring every half-hour or so, and lasting one day". GP costs of €44 should be added, giving a total of €84.

Note the endpoint here is asthma related GP visits – not new cases of asthma. The latter has higher costs. In this context, there has been recent work in the UK (HSE, 2003) which estimated the cost of a new case of asthma at between £42,000 and £45,000 (about €60,000). These costs include: loss of income through absence from work or having to change jobs; medical treatment; and pain and suffering.

4.10.5 Restricted activity days (RAD)

As stated in the section on quantification, RADs - in order of decreasing severity, - include: days when a person needs to stay in bed, days when a person stays off work or school (or whatever may be their usual place to go, if they have a usual place to go) but doesn't need to stay in bed, other, less serious, restrictions on normal activity. (These are what are called 'minor' RADs, valued separately below.)

A WTP value of €154 is available from the Ready et al. (2004) study. Here, the symptom is described as three days confined to bed, where there is shortness of breath on slight exertion. This description matches well the most severe definition of a RAD above. To this it is necessary to add loss of productivity. The proportion of RADs and mRADs were averaged in line with the assumptions in section 10.2.2 (assuming 25%:25%:50%: for working adults between days in bed, work loss days, and mRADs). A best estimate of €92 was used in EC4MACS.

4.10.6 Respiratory symptoms in people with asthma: Adults and children

The asthma attack values given above for adult asthmatics - €145 per event and €16 per extra day - may be used. For asthma attacks among the respondents' own children the WTP per event was €307, and a WTP of €31 for each additional day of asthma symptoms. The value of €40 used for lower respiratory symptoms may be used instead but it is judged that the asthma value, whilst not the end-point being valued, allows us to consider the WTP values of people who suffer regularly from a similar condition. All these values are derived from the Ready et al. (2004) study.

4.10.7 Respiratory medication use by children and adults

Regular use of respiratory medication includes the use of bronchodilators. The resource costs of drugs typically associated with bronchodilators vary between €0.5 and €1 per day, according to whether Terbutaline or Albuterol is used¹². We do not have any evidence for the value of disutility of using bronchodilators and so factor this in implicitly by assuming the total unit value for these end-points is at the upper end of the range presented above, i.e. €1 per day. We do not differentiate between children and adults since use rates of bronchodilators – and therefore unit costs - are assumed to be the same for both groups.

¹² <http://www.fpnotebook.com/LUN118.htm>

4.10.8 Chronic bronchitis (new cases)

We stress that there are few studies on air pollution and development of chronic bronchitis. There are questions about whether a study that is often used for quantification of this endpoint, that by Abbey et al. (1995), is sufficiently representative and suitable for use in European HIA. Again, there are issues with definition of chronic bronchitis and with baseline rates of incidence. However, results from other studies show that when quantified and monetised, what seem to be the pollution-attributable new cases of chronic bronchitis can have a substantial impact on final benefits estimates.

To our knowledge no primary empirical research has been undertaken in the EU to derive unit values for new cases of chronic bronchitis. We are therefore forced to rely on the results of studies undertaken elsewhere. The two studies that we have reasonable confidence in – at least with regard to their methodological robustness – are those by Viscusi et al (1991) and Krupnick and Cropper (1992), which both use a survey (CVM) approach. We base our selection of unit values on the results of these studies. However, we must highlight the fact that transferability of the results from these studies may be limited by their being i) undertaken in the US, and ii) dated by 15 years. Their use is further circumscribed by the fact that the definitions of chronic bronchitis used in the studies do not coincide with those used in the epidemiological studies that attempt to quantify the number of cases due to air pollution. We discuss ways of dealing with these issues in the following paragraphs.

The two valuation studies define chronic bronchitis consisting of the following health features:

- Living with an uncomfortable shortness of breath for the rest of your life
- Being easily winded from climbing stairs
- Coughing and wheezing regularly
- Suffering more frequent deep chest infections and pneumonia
- Having to limit your recreational activities to activities such as golf, cards, and reading
- Experiencing periods of depression
- Being unable to do the active, physical parts of your job
- Being limited to a restricted diet
- Having to visit your doctor regularly and to take several medicines
- Having to have your back mildly pounded to help remove fluids built up in your lungs
- Having to be periodically hospitalized
- Having to quit smoking
- Having to wear a small portable oxygen tank

It is recognized by both sets of authors that this description constitutes the most severe definition of the chronic bronchitis end-point. The Krupnick and Cropper study therefore attempts to scale these symptoms by asking their survey respondents who had relatives with chronic bronchitis to rank their relatives' illnesses against the "severe" case definition on the basis of the number and type of symptoms present. This is a similar approach to that adopted by Maddison (1997) where he plots different health end-points associated with air pollution on a Quality of Well Being index and then derives WTP by scaling (linearly) from an established WTP for an end-point. Using the evidence from their WTP questions and the scaling exercise, the authors were able to estimate that the WTP for an average case of chronic bronchitis was 58% lower than that for the severe case.

We can combine this WTP-scaling result with the WTP estimates for the severe case as estimated by the Viscusi et al. study (this study being the better in terms of representative sampling and size). The Viscusi et al. study derives WTP for chronic bronchitis in two principal ways:

- WTP - Risk of CB trade-off
- Risk of CB – Risk of car accident fatality trade-off

The WTP - Risk of CB trade-off method produces a value of \$504,500, or €698,400. The Risk of CB – Risk of car accident fatality trade-off is put at 0.32 – that is, a case of severe chronic bronchitis is valued at 32% of an accident VSL. Thus, if the accident fatality is valued at €1 million (after Carthy et al, 1999) the case of severe chronic bronchitis would be valued at €320,000. Using these two values to derive a value range, we may use a mid-point of €476,300 as a central estimate. Applying the scaling factor of 0.42 from the Krupnick and Cropper study we can derive the following values to be used in the current context:

High range estimate:	€276,600
Central range estimate:	€208,200
Low range estimate:	€139,900

The validity of using these values in EC4MACS first depends on our assuming that the average severity of a case of chronic bronchitis found in the Krupnick/Cropper study is close to how it is defined in the epidemiological literature. If this assumption is accepted, and we accept that the direct temporal and spatial transfer is valid then we can adopt this range as an indicative range.

4.10.9 Other end-points

The Ready et al. (2004) study also notes that one cough day is estimated to be €42/day. The same value should be applied to minor RAD (restricted activity day) and symptom day (note that this is probably a low estimate for a symptom day as one day with mildly, red watering, itchy eyes and runny nose is valued at €56).

4.10.10 Summary of health endpoints

The table below summarises the values that have been used in the current study. For mortality several values are listed for sensitivity analysis. It is intended that in the near future there will be a rationalisation of these figures.

Table 8. Summary of mortality and morbidity unit values.

Health end-point	Recommended central unit values, € price year 2005
Value of a life year (VOLY)	57,700 / 138,700 40,000 (Desaigues et al sensitivity)
Value of statistical life (VSL)	1.09 million / 2.22 million 2.8 million (OECD sensitivity)
Infant mortality	1.64 million / 3.33 million 4.2 million (OECD sensitivity)
Hospital admissions	2,260/admission
GP visits (event): Asthma Lower respiratory symptoms	59/consultation 84/consultation
Respiratory symptoms in asthmatics (event): Adults Children	144/event 309/event
Respiratory medication use – adults and children (day)	1/day
Restricted activity day (adjusted average for working adult)	92/day
Restricted activity day (adjusted average for age >65)	75/day
Restricted activity day (days when a person needs to stay in bed)	143/day
Restricted activity day (work loss day)	139/day
Minor restricted activity day	42/day
Cough day	42/day
Symptom day	42/day
Work loss day	130/day
Minor restricted activity day	42/day
Chronic bronchitis	208,000/case

5. Agricultural and Horticultural Production

5.1. Introduction

This impact category is referred to here as ‘agricultural and horticultural production’ to recognise that, potentially, there could be effects on livestock production as well as crop production, even though there has been little attempt in the past to quantify the former. Impacts mapped out by pollutant are identified below. Some air pollutants other than those listed have been linked in the literature to crop damages, particularly HF, though European emissions are now low because of modern industrial pollution controls.

Table 9. Impacts of the principal CAFE air pollutants on agricultural and horticultural production

Pollutant	Effect
SO ₂	<ul style="list-style-type: none"> • Increase in yield at low exposure, decrease in yield at high exposure • Visible injury at high concentrations (would make some leaf crops such as spinach or lettuce un-saleable) • Enhanced performance of pests and pathogens • Acidification of agricultural soils
NO _x	<ul style="list-style-type: none"> • Enhanced performance of pests and pathogens • Reduced tolerance of other stresses (e.g. drought, cold) • Acidification of agricultural soils • Fertilisation of agricultural systems with nitrogen • Increased nitrogen run-off from agricultural systems
NH ₃	<ul style="list-style-type: none"> • Enhanced performance of pests and pathogens • Reduced tolerance of other stresses (e.g. drought, cold) • Acidification of agricultural soils • Fertilisation with nitrogen • Increased nitrogen run-off from agricultural systems
Ozone	<ul style="list-style-type: none"> • Visible injury to crops • Reduction in crop yield • Enhanced performance of pests and pathogens • Interaction with climate
Particles	<ul style="list-style-type: none"> • Some discussion in the literature of impacts through shading effects of particles deposited on leaf surfaces • Decrease in photosynthetically active radiation reaching plants

Past analysis (e.g. the ExternE national implementation reports summarised in ExternE, 1999b) suggests that a number of these effects are now likely to be unimportant. A major reason for this is that the past few years have seen significant falls in emissions of SO₂ and other pollutants, such that in most agricultural areas concentrations are now well below those observed to cause damage. Largely in view of this the following are not considered in detail in EC4MACS:

- Direct effects of SO₂, NO_x, NH₃ and particles on yield;
- Visible injury caused by SO₂;
- Acidification of agricultural soils (which could be quantified through consideration of additional liming requirements for soils, noting that farmers routinely apply lime to counteract acidification linked to harvest);

- Fertilisational effects of deposited sulphur and nitrogen (which for nitrogen could be quantified in terms of the cost equivalent of N-fertiliser);
- Effects of nitrogen run-off.

Stakeholders were invited under CAFE to comment on the exclusion of these effects, noting that the list contains some balance between potentially positive impacts (e.g. associated with nutrient inputs to agricultural land) and negative impacts (e.g. soil acidification and nutrient runoff). No comments were received, from which it was concluded that there is general agreement with this position.

The following effects of air pollution on agriculture have been identified for specific consideration within the benefits assessment (Table 10):

Table 10. Impacts for detailed quantitative or qualitative consideration in EC4MACS, extent of analysis depending on the availability of data

<p>Ozone only</p> <ul style="list-style-type: none"> • Visible injury to crops • Reduction in crop yield • Interaction with climate • Reduction in livestock production 	<ul style="list-style-type: none"> • Qualitative discussion • Quantitative assessment • Qualitative assessment • Qualitative discussion
<p>All pollutants</p>	<ul style="list-style-type: none"> • Qualitative discussion of interactions with pests and pathogens

Ozone is recognised as the most serious regional air pollutant problem for the agriculture and horticulture sectors in Europe at the present time. The effects of air pollution, particularly ozone, on crop yield have been quantified in a number of papers over the years in Europe (van der Eerden et al, 1988; AEA Technology, 1999; Holland et al, 2002) and in the USA (e.g. Adams et al, 1985; Shortle et al, 1986; Olszyk et al, 1988). Interestingly there was little or no cross-over in the functions used between the American and European studies, reflecting the existence of a substantial effects literature on both sides of the Atlantic in this field. However, Hornung and Jones (1999) used American data to modify a European function developed for spring wheat according to the relative sensitivity of a number of crops for which there is little or no European data.

As will be shown, methods for quantification of ozone effects on productivity are still under development, though sufficient knowledge is now available to enable a reasonable estimate of its direct impacts to be made. In other areas (visible injury, interactions between pollutants and pests and pathogens, impacts on livestock production), quantification has not been practicable on the European scale, though research is continuing.

5.2. Quantification of the direct effects of ozone on yield

The quantification needs the following data:

1. Information on stock at risk, in terms of the distribution of crop production, by species, across Europe.
2. Exposure-response functions for different crops, recognising the variability in response between species.

3. Valuation data.

5.2.1 Stock at risk data

The following describes the information on crop production currently held in ALPHA2, and generated at SEI-York. The SEI-York land cover map is made up of discrete data layers (forests, semi-natural vegetation, urban, water bodies etc.) with each data layer being created by combining various existing land cover data sets. For the delimitation of agricultural areas and their linkage to the crop production statistics, the following data layers were combined: IGBP Global Land Cover (GLC) agricultural information; SEI Land Cover agricultural information and the Bartholomew Country and NUTS region boundaries

The GLC classification of agricultural classes was the dominant data source used to spatially delimit the distribution and type of crop lands across Europe. Firstly, the areas which were purely agricultural were identified, for example, "Cropland (Winter Wheat, Small Grains)". Next, the areas that were of mixed classes combined with forestry were delimited, for example, "Cropland (Rice, Wheat) with Woodland". Thirdly, areas that were mixed classes of agriculture, forestry and grassland were identified, for example, "Cropland and Pasture (Wheat, Orchards, Vineyards) with Woodland". The SEI agriculture map was then used to differentiate the extent of cropland from pasture and the distribution and classification of horticultural areas. The reclassified land use map contained approximately 250 discrete classes.

In order to combine the spatial database with statistical crop information, the agricultural map was overlaid with data sets showing the distribution of country boundaries, distribution of European NUTS level II areas and the EMEP 50km grid. This produced a database onto which country and NUTS specific information on yield and crop coverage could be appended and the results analysed by EMEP grid square. This data was then combined with statistical information from the EUROSTAT Agricultural Statistics for EU NUTS Level II and the FAO AGROSTAT Agricultural Statistics for the remaining European Countries.

For each country a specific database of the percentage coverage of crops and yields linked to the agricultural map was produced using the FAO agrostat data. For example, the grain class from the FAO map was defined as wheat, barley, rye, oats and millet with small grains being the subset of wheat, barley, rye and oats. The breakdown of the split of crop types by country in each polygon of the agricultural map was then used to calculate the area and yield of crops in each EMEP 50km grid square. A similar activity was performed for NUTS Level II regions for those countries that had reported yields and crop areas for 1999. The countries included in this more detailed disaggregation were the UK, Italy, France, Germany, Finland, Belgium and Luxembourg. The final stage of the mapping exercise required correction of estimated total yields for each country against total annual yields as reported to FAO.

5.2.2 Response functions

European experimental work has generated a number of response functions based around the AOT40 metric ozone (AOT40 being the sum of hourly concentrations in excess of 40 ppb during the day [‘day’ defined as those hours with a clear sky global radiation of 50 Wm^{-2} or more] over the 3 month period with the highest running sum ozone concentration). Despite this, quantification remains controversial (Karenlampi and Skarby, 1996, and others since, have stated that AOT40 can be used to set a critical level, but not be taken further to

quantification). The reason for this is that the AOT40 functions do not take account of the potential for plant response to a given level of pollution to change in response to other factors, such as the age of plants, the presence of other pollutants, time of day, temperature, water status and humidity, soil and plant nutrient status, species and cultivar, presence of pests and pathogens, and possibly other factors (ExternE, 1995). This is partly due to the fact that AOT40 is not a direct estimate of ozone actually absorbed by the plant. Accounting for this requires use of a model simulating ozone absorption by a vegetation canopy. Scientists working in this area have for some time been developing flux-based approaches that seek to account for some of these interactions, and these functions are beginning to emerge. ICP/MM (2004) Mapping Manual provides the following functions:

$$RY_{\text{wheat}} = 1.00 - (0.048 * AF_{\text{st}6}) \quad [\text{wheat, based on data from Finland, Belgium, Sweden and Italy, see Figure 1}]$$

$$RY_{\text{potato}} = 1.01 - (0.013 * AF_{\text{st}6}) \quad [\text{potato, based on data from Finland, Sweden, Belgium and Germany, see Figure 2}]$$

RY = Relative yield

$AF_{\text{st}6}$ = Accumulated stomatal flux of ozone over a threshold of 6 $\text{mmol} \cdot \text{m}^{-2}$.

There is debate as to what ‘threshold’ figure should be used to provide the most reliable relationship between exposure and impact on yield.

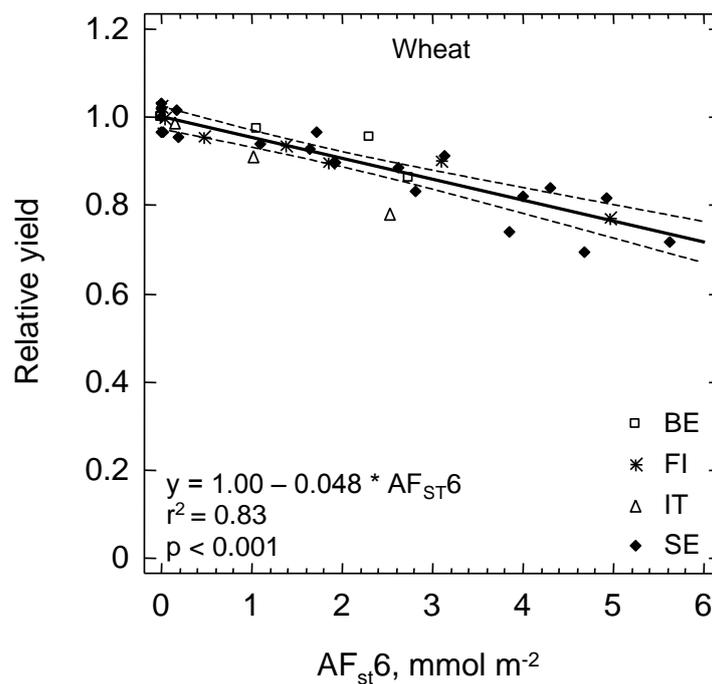


Figure 5. The relationship between the relative yield of wheat and $AF_{\text{st}6}$ for sunlit leaves in wheat based on five wheat cultivars from four European countries using effective temperature sum to describe phenology.

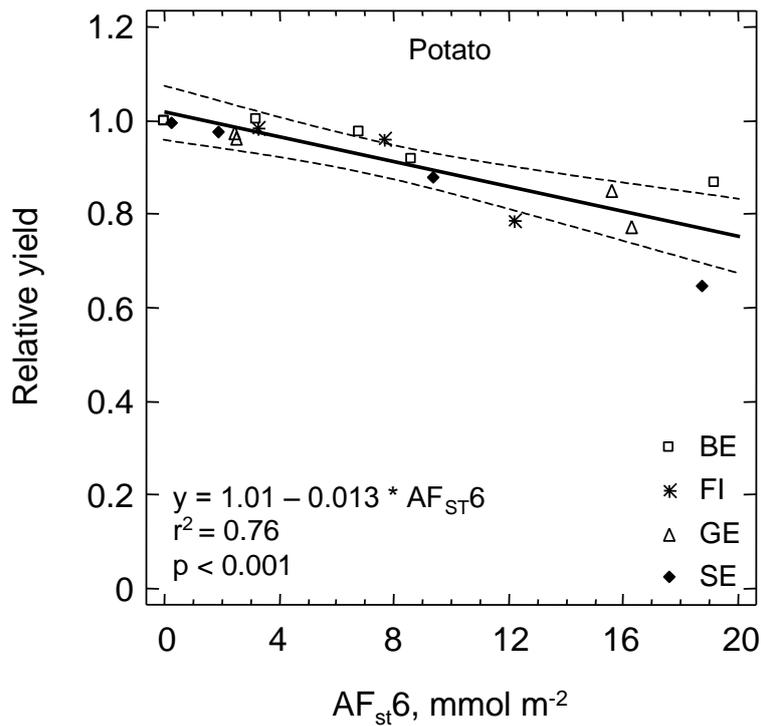


Figure 6. The relationship between the relative yield of potato and the AF_{st6} for sunlit leaves based on data from four European countries and using effective temperature sum to describe phenology.

Although wheat and potato are very important crops, together they comprise only about 10% of European agricultural production. It is thus clearly desirable that a more comprehensive analysis is undertaken. There are two ways of doing this, drawing on the flux modelling that is currently possible:

1. Adaptation of the flux based functions for wheat and potato based on relative sensitivity of other crops according to available AOT40 data. Table 11 reproduces one view of relative sensitivity, taken from Mills et al (2003) and cited in the mapping manual (ICP/MM, 2004).
2. Quantify using AOT40 functions modified between European regions in line with variation seen in flux-based assessments for wheat and potato.

Table 11. The range of sensitivity of agricultural and horticultural crops to ozone with reference to AOT40 data (see Mills et al, 2003, 2006 for response functions)

Sensitive (Critical Level ≤ 5 ppm.h)	Moderately sensitive (Critical Level of 5 – 10 ppm.h)	Moderately resistant (Critical Level of 10 - 20 ppm.h)	Insensitive (Critical Level >20 ppm.h)
Cotton, Lettuce, Pulses, Soybean, Salad Onion, Tomato, Turnip, Water melon, Wheat	Potato, Rapeseed, Sugarbeet, Tobacco	Broccoli, Grape, Maize, Rice	Barley, Fruit (plum & strawberry)

During the EC4MACS study it was not possible to integrate flux measurements with the GAINS modelling. Accordingly, estimates of changes in crop production were based on AOT40 methods. However, once flux estimates are available the benefits assessment will use those data, possibly linked to other European models (specifically work by Mills and colleagues at CEH, UK).

5.2.3 Valuation data

Past analysis has taken a simplistic approach to valuation of impacts on agricultural production, with estimated yield loss being multiplied by world market prices as published by the UN's Food and Agriculture Organization. The values used in CAFE are shown in Table 12. World market prices were used as a proxy for shadow price on the grounds that they are less influenced by subsidies than local European prices (in other words, they are closer to the 'real' price of production). A similar approach has been followed in EC4MACS. Whilst air pollution overall may be capable of changing production sufficiently to affect prices, it is arguable whether marginal change associated with specific policies, would be sufficient to do so to any noticeable degree.

Table 12. Valuation data for agricultural crops. All data taken from the FAO website, 2000.

	€/tonne		€/tonne
Barley	120	Pulses	320
Carrots	340	Rape	240
Cotton	1350	Rice	280
Fresh vegetables	340	Rye	80
Fruit	680	Soya	230
Grape	360	Sugar beet	60
Hops	4100	Sunflower	240
Maize	100	Tobacco	4000
Millet	90	Tomato	800
Oats	110	Water melon	140
Olives	530	Wheat	120
Potatoes	250		

6. Materials

6.1. Types of impact

Air pollution is associated with a number of impacts on materials:

- Acid corrosion of stone, metals and paints in ‘utilitarian’ applications;
- Acid impacts on materials of cultural merit (including stone, fine art, and medieval stained glass, etc.);
- Ozone damage to polymeric materials, particularly natural rubbers;
- Soiling of buildings and materials used in other applications.

Of these, monetary quantification has been carried out for all groups in previous studies, though not necessarily at the European scale. In particular, economic assessments of impacts on cultural heritage are very limited. An alternative approach developed by ICP Materials for informing decision makers is based on assessment of ‘acceptable rates’ of deterioration, on the basis that materials left in the open air will be damaged even in the absence of air pollution. This approach is discussed in more detail below, though of course the concept of an ‘acceptable rate’ begs the question of what is acceptable. It is also a measure that cannot be integrated with a CBA other than through bias analysis in the uncertainty assessment. It may also require data at a far higher level of resolution than is yet available for anything other than extremely local modelling if it were able to pick up changes in rate from specific policies.

6.2. Acid damage to materials in ‘utilitarian’ applications

This section deals with damages to materials in utilitarian applications, in other words, those used in buildings, etc., of no special cultural merit. This would include most modern houses, factories, railway buildings, etc.

The following text is taken from the website of ICP Materials and describes trends in damage, which clearly show benefits of existing action on air pollution in Europe. The trend exposure consisted of repeated 1-year exposures of steel and zinc on 39 test sites during the period 1987-1995. Of the environmental parameters investigated only concentrations of SO₂, NO₂ and H⁺ (acidity) exhibit trends. All of these are decreasing, with SO₂ having the strongest trend and NO₂ the weakest. For O₃ no specific trends were observed. The decreasing trend in the concentration of acidifying air pollutants has resulted in decreasing corrosion rates of the exposed materials. Both carbon steel and zinc show reduced corrosion rates in unsheltered as well as in sheltered positions, with changes in SO₂ levels being the largest single contributing factor to the trends. The decreasing acidity of precipitation is a contributing factor, though its effect is much smaller than that of dry deposition. The decrease in corrosivity is generally larger than expected from the drop of SO₂ and H⁺ concentrations (reflecting uncertainty in exposure-response relationships). This part cannot be directly related to a specific pollutant and reflects the multi-pollutant character of process of material degradation.

6.2.1 Framework for assessment

Assessment of acid damage to building materials in everyday applications uses the following framework, which is broadly similar to that described elsewhere in this paper:

1. Describe the stock-at-risk in terms of the exposed area of sensitive materials on buildings and other structures within each country;
2. Describe background pollution levels (important because of non-linearities in exposure-response);
3. Describe climatic variables relevant to the response functions;
4. Apply material-specific response functions that provide estimates of the rate of corrosion, etc.;
5. Assess rate of deterioration against a critical loss of material to define change in maintenance or replacement frequency;
6. Value using standard maintenance cost data from reference sources used by builders and architects.

6.2.2 Stock at risk data

The stock at risk is derived from data on building numbers and construction materials taken from building survey information. The information listed here was collated for the EC ExternE Project (ExternE, 1995, 1999a). Inventories are generally developed for individual cities; these can then be extrapolated to provide inventories at the national level (accepting the uncertainties inherent in the extrapolation). For countries for which data are not available, values must be extrapolated from elsewhere, though this inevitably results in lower accuracy. In general it is assumed that the distribution of building materials (m^2) follows the distribution of population (i.e. m^2 /person). Sources of data are as follows:

Eastern Europe (including the eastern Lander of Germany):

Kucera *et al* (1993a,b), Tolstoy *et al* (1990) - data for Prague

Scandinavia:

Kucera *et al* (1993 a,b), Tolstoy *et al* (1990) - data for Stockholm and Sarpsborg

UK, Ireland:

Ecotec (1996), data for UK extrapolated to Ireland

Western Lander of Germany:

Hoos *et al* (1987) - data for Dortmund and Köln

Other western Europe:

Average of material use per person from Hoos *et al*, Kucera *et al* and Tolstoy *et al* (excluding Prague), and Ecotec.

For galvanised steel in structural (non-building) applications an average of material data was derived from European Commission (1995) and Kucera *et al* (1993 a,b).

6.2.3 Exposure-response functions

The following series of exposure-response functions is taken from the ICP Materials (2003) website. Functions are based on exposures over an 8 year period (1987 to 1995). The most important for the CAFE assessment are likely to be those relating to steel, zinc and stone

(limestone and sandstone), and application for mortar and rendering. The ICP has generated other functions, for example for copper, cast bronze some types of glass and some electric contact materials. These are not reproduced here as they are of limited relevance to utilitarian material applications, though they are of course important for consideration of effects on cultural heritage. Finally, there are also functions for paint damage, though previous work (e.g. in ExternE) has found the use of these functions difficult. Given changes in paint formulation over time (particularly the move from paints based on organic solvents to water based paints) it is questionable whether the functions identified for paint are relevant any more.

Key:

ML = Mass loss (g/m^2)

$\text{SO}_2, \text{NO}_2, \text{O}_3$ = concentration in $\mu\text{g.m}^{-3}$

T = temperature ($^{\circ}\text{C}$)

Rain = precipitation (mm/year)

H^+, Cl^- = hydrogen and chloride ion concentration in rainfall (mg/l)

R = surface recession (μm)

Rh = relative humidity (%)

t = time in years

ASTM is the degradation of coatings measured according to ASTM D 1150-55, 1987, giving a range between 1 and 10 where 10 corresponds to an unexposed sample.

Unsheltered Exposure

Weathering steel (N=148, $R^2=0.68$)

$$\text{ML} = 34[\text{SO}_2]^{0.33} \exp\{0.020\text{Rh} + f(\text{T})\}t^{0.33}$$

$$f(\text{T}) = 0.059(\text{T}-10) \text{ when } \text{T} < 10^{\circ}\text{C}, \text{ otherwise } -0.036(\text{T}-10)$$

Zinc (N=98, $R^2=0.84$)

$$\text{ML} = 1.4[\text{SO}_2]^{0.22} \exp\{0.018\text{Rh} + f(\text{T})\}t^{0.85} + 0.029\text{Rain}[\text{H}^+]t$$

$$f(\text{T}) = 0.062(\text{T}-10) \text{ when } \text{T} < 10^{\circ}\text{C}, \text{ otherwise } -0.021(\text{T}-10)$$

Aluminium (N=106, $R^2=0.74$)

$$\text{ML} = 0.0021[\text{SO}_2]^{0.23} \text{Rh} \cdot \exp\{f(\text{T})\}t^{1.2} + 0.000023\text{Rain}[\text{Cl}^-]t$$

$$f(\text{T}) = 0.031(\text{T}-10) \text{ when } \text{T} < 10^{\circ}\text{C}, \text{ otherwise } -0.061(\text{T}-10)$$

Portland limestone (N=100, $R^2=0.88$)

$$\text{R} = 2.7[\text{SO}_2]^{0.48} \exp\{f(\text{T})\}t^{0.96} + 0.019\text{Rain}[\text{H}^+]t^{0.96}$$

$$f(\text{T}) = -0.018\text{T}$$

White Mansfield sandstone (N=101, $R^2=0.86$)

$$\text{R} = 2.0[\text{SO}_2]^{0.52} \exp\{f(\text{T})\}t^{0.91} + 0.028\text{Rain}[\text{H}^+]t^{0.91}$$

$$f(\text{T}) = 0 \text{ when } \text{T} < 10^{\circ}\text{C}, \text{ otherwise } -0.013(\text{T}-10)$$

Coil coated galvanised steel with alkyd melamine (N=138, $R^2=0.73$)

$$(\text{10-ASTM}) = (0.0084[\text{SO}_2] + 0.015\text{Rh} + f(\text{T}))t^{0.43} + 0.00082\text{Rain} \cdot t^{0.43}$$

$$f(\text{T}) = 0.040(\text{T}-10) \text{ when } \text{T} < 10^{\circ}\text{C}, \text{ otherwise } -0.064(\text{T}-10)$$

Steel panels with alkyd (N=139, $R^2=0.68$)

$$(\text{10-ASTM}) = (0.033[\text{SO}_2] + 0.013\text{Rh} + f(\text{T}))t^{0.41} + 0.0013\text{Rain} \cdot t^{0.41}$$

$$f(\text{T}) = 0.015(\text{T}-11) \text{ when } \text{T} < 11^{\circ}\text{C}, \text{ otherwise } -0.15(\text{T}-11)$$

Exposure under a sheltering roofWeathering steel (N=148, R²=0.76)

$$ML = 8.2[SO_2]^{0.24} \exp\{0.025Rh + f(T)\}t^{0.66}$$

$$f(T) = 0.048(T-10) \text{ when } T < 10^\circ\text{C, otherwise } -0.047(T-10)$$

Zinc (N=91, R²=0.80)

$$ML = 0.058[SO_2]^{0.16} Rh \cdot \exp\{f(T)\}t^{0.49}$$

$$f(T) = 0.039(T-10) \text{ when } T < 10^\circ\text{C, otherwise } -0.034(T-10)$$

The exposure-response functions require data on meteorological conditions. Of these, the most important are precipitation and humidity.

To convert mass loss (ML) for metals into an erosion rate in terms of material thickness, simply requires knowledge of the density of the materials concerned (e.g. for zinc a density of 7.14 tonnes/m³ can be assumed).

A few issues are relevant for implementation:

- Dose-response functions for stone materials are based on a combination of functions from sheltered and unsheltered conditions. These reflect different deterioration mechanisms. We only apply dose-response functions for unsheltered conditions. For sheltered conditions the possibility of including soiling effects is investigated in a later section. The inventory therefore needs to make an estimate of the proportion of unsheltered material.
- Based on observation it is assumed that some common building materials are unlikely to be affected by air pollution, these being brick and aluminium. The reference to brick does not include surrounding mortar, which is considered sensitive.
- ICP Materials functions only exist for limestone, sandstone and zinc. Other materials (mortar, rendering, and galvanised steel) are expected to behave similarly to sandstone or zinc.

Given rather limited European data on the response of paints to acid deposition, past work on the NEC Directive used functions from a US source (Haynie, 1986) to assess damage to carbonate- and silicate-based paints. Given the age of these functions, and the rate at which paint technology has developed in the last 20 years, it is not clear how relevant it is to Europe a quarter of a century on from the time that the functions were published (as noted above). The ICP analysis includes two functions for paint (Coil coated galvanised steel with alkyd melamine). The critical point for the paint coating material is when ASTM=5. However, in order to implement these functions, an estimate is needed of the baseline status of material. Previous implementations have therefore not progressed to quantification.

6.2.4 Valuation

Valuation of the loss of material requires some assumption to be made about human behaviour in relation to maintenance. In past work (e.g. ExternE, 1995, 1999a; AEA Technology, 1999) it has been assumed that the owners of buildings and other structures are perfectly rational and undertake maintenance once a 'critical thickness' of material has been lost (corresponding to the time when it is most economically appropriate to undertake repair or maintenance), and prior to secondary damage (e.g. wood rot in window frames) becomes a problem. Assumptions on the critical thickness loss for maintenance and repair of different materials are as follows:

Table 13. Assumed critical thickness for maintenance or repair measures for building materials.

Material	Critical thickness loss
Natural stone	3-5 mm
Rendering	3-5 mm
Mortar	3-5 mm
Zinc:	
Construction - sheet and strip	25 μm
Other construction, agriculture and street furniture	50 μm
Pylons, other transport	100 μm
Galvanised steel	15 - 120 μm
Paint	20 - 100 μm

Critical thicknesses were quantified as follows:

- For zinc, galvanised steel and paint, from information on the thickness of the coating.
- For stone and stone-based materials, from consideration of the average amount of degradation across a surface that would necessitate repairs being made. On any surface of any significant size it is likely that some areas suffer much heavier damage than others, so implicit within the 3-5 mm average is an assumption of degradation by 1 cm or more in some parts. This is sufficient to prompt most home-owners to undertake repairs, such as repointing the mortar-work between bricks.

Valuation data, derived using reference materials used by the building trade, are given below:

Table 14. Repair and maintenance costs [2005€/m²] proposed for this analysis.

Material	€/m²
Zinc	25
Galvanised steel	30
Natural stone	280
Rendering, mortar	30
Paint	13

Uncertainties in the quantification

A series of uncertainties are present in the proposed analysis, of which the following appear likely to be the most important:

1. Development of the inventory of stock-at-risk, involving extrapolation of data from a number of studies in specific cities;
2. Application of a limited number of response functions to materials that will vary in some ways from the experimentally exposed samples;
3. Extrapolation of response data from small samples to materials used on buildings which will differ in their exposure characteristics;
4. Determination of the critical thickness for the different materials;
5. Assumption that building (etc.) owners react to material damage in a purely rational manner.

6.2.5 Approach followed in EC4MACS

The functions, etc. used above have previously been applied in the estimation of marginal damage costs (€/tonne of pollutant) for SO₂. Inclusion of other pollutants made little difference to the results. Results demonstrate that associated damage is very small, 1% or less of the damage linked with health impacts. Further to this, the GAINS model does not routinely generate the range of pollutant metrics required by the materials response functions. Whilst the full modelling framework for utilitarian materials has been retained, it is not integrated with ALPHA or ALPHA-Riskpoll. However, an indication of likely results is provided using the outputs of the marginal damage assessments referred to above.

6.3. Acid damage to cultural heritage

Corrosion of cultural heritage from acid rain leads to irreversible damage, as restoration will never be able to return the undamaged originals. Against this, materials exposed to even an unpolluted atmosphere will degrade over time, but at a much slower rate.

Quantification based on the impact pathway approach as used for acid damage to utilitarian buildings is not possible for cultural heritage for the following reasons:

- An appropriate stock-at-risk does not exist for European cultural heritage;
- Critical thickness for damage will vary considerably;
- Valuation will also vary considerably.

Given that it is not possible here to provide an economic evaluation of damage to cultural heritage, it is necessary to consider how it might be integrated with the benefits assessment in the future. The omission of these effects can clearly give a downward bias to the results (though the extent of this will vary significantly depending on which pollutants are being controlled), so it is appropriate to say as much when comparing costs and benefits. To elaborate on that it would be possible to estimate the rate of damage of stone and other relevant materials, and perhaps band major European cities in which cultural heritage is most concentrated into different risk categories. Sufficient data are available to do this already, for example, in relation to work by ICP materials in the context of the revision of the Gothenburg Protocol.

Navrud, in Holland et al (2005a) provided a summary of the relevant literature on valuation of cultural heritage drawing on Navrud and Ready (2002), to which readers should refer for more information.

6.4. Ozone damage to polymeric materials

Although ozone is a major determinant of the lifetime of many rubber materials exposed to the ambient air, we know of only two studies in Europe that have investigated the problem from an environmental perspective. Lee et al (1996) estimated annual damages to the UK of £170 to £345 million (GB pounds) for impacts on surface coatings (paints) and elastomers and the cost of anti-ozonant protection used in rubber goods. These estimates were based on US data from the late 1960s, demonstrating the dearth of information in this area. Lee's work served as a scoping study for a larger project (Holland et al, 2007) that undertook experimental assessments of a range of paints (96 in all, selected to be representative of the UK market) and rubber formulations.

Various effects from ozone exposure were noted on the paint samples, including colour change, change in flexibility, resistance to petrol and water, and changes in gloss. There was no clear relationship between these changes – for example, the fact that a sample showed one effect did not mean that it would be more likely to show any of the other changes. A dose-response function was developed for colour change, which was the most common effect. Application of this function gave the result that on average most paints that were sensitive would not change colour noticeably (when compared to original samples) until about 12 years had passed. As this is longer than the service lifetime of most paints it was concluded that the direct effects of total ozone exposure on paints had no significant economic cost in the UK. The effects on paints of a marginal change in ozone levels would thus also be negligible. The possible effects of interactions of ozone with other environmental stresses in damaging paints were not addressed.

The work on rubber goods required development of an extensive inventory of rubber products and use for the UK. The inventory was divided into those products that would be ozone-sensitive and those that would not be (e.g. products likely to be made from some synthetic rubbers). A dose-response function was generated for crack growth in sensitive materials, allowing assessment of the lifetime of products. These data were used to estimate total annual damages to rubber goods in the UK of £35 to 189 million, with a best estimate of £85 million/year. The effect of a 1 ppb change in ozone was estimated at £4 million/year.

This latter estimate could be used to make a rough estimate of ozone damage to rubber products for EC4MACS. Whilst it would not be possible to develop inventories for the whole of Europe under this project, it may be appropriate to extrapolate the UK result based on the size of each country's vehicle fleet (the motor industry dominates rubber product utilisation) using the following expression:

$$\text{Damage/ppb_change_}O_3 = \text{£4million} \times \frac{\text{vehicles_in_country_}x}{\text{vehicles_in_UK}}$$

The nature of this extrapolation is clearly rather crude, though the magnitude of damage relative to those from health, etc., mean that associated uncertainties have a trivial effect on the results.

6.5. Soiling of buildings

Soiling of buildings by particles is one of the most obvious signs of pollution in urban areas. Soiling affects both 'utilitarian' and historic buildings and causes economic damages through cleaning and amenity costs. Particles may also be involved in damage to building fabric directly¹³.

Soiling is an optical effect (a darkening of reflectance), caused primarily by the deposition of airborne particulate matter onto external building surfaces. The factors which can affect the degree of soiling include (QUARG, 1996): the blackness per unit mass of smoke; the particle size distribution; the chemical nature of the particles; substrate-particle interfacial binding; surface orientation; and micro-meteorological conditions. Similarly, different types of particulate emission have different soiling characteristics, shown in the table below. They can

¹³ Carbon particles may play a role as a catalyst for stone erosion, particularly in the conversion of SO₂ and NO_x into sulphuric and nitric acids

be differentiated by fuel type with the use of dark smoke emission factors (Newby *et al*, 1991; Mansfield *et al*, 1991). For example, diesel emissions have a much higher soiling factor than petrol or domestic coal emissions. This is due to their particulate elemental carbon (PEC) content (QUARG, 1993). PECs have a high optical absorption coefficient and their hydrocarbon content means they are very sticky and much less water soluble than suspended soil particles (which are readily removed by rain washing; Mansfield, 1992). Therefore, a PEC particle landing on a surface is more likely to strongly adhere than other particulate matter. Diesel emissions are the main source of atmospheric PEC in Western Europe.

Table 15. Smoke emissions and relative soiling factors

Fuel	smoke emission factor (% by mass)	dark smoke emissions factor (by mass)	Soiling factor relative to coal
Coal (domestic)	3.5	3.5	1.0
Diesel	0.6	1.8	3.0
Petrol	0.15	0.065	0.43

Source: Mansfield, 1992.

Although soiling damage has an obvious cause and effect, the quantification of soiling damage is not straightforward. Soiling can impact on a number of different materials, including natural stone, paint, concrete, rendering and also potentially glass. The latter effect may be important, though there are limited studies investigating the potential effects¹⁴. Most analysis to date has been undertaken on stone buildings. These studies show through measurement data on reflectance (and industry experience) that soiling appears to be very rapid on clean surfaces, following initial exposure. The effect is therefore strongly non-linear. Moreover, evidence shows that reflectance measurements oscillate, indicating a cleansing and re-soiling process. This may occur as soil derived particles may be deposited on materials but are more likely to be removed by rainfall than a deposited diesel particle.

There are a number of dose-response functions in the literature for soiling. It is possible to proceed to valuation, based on cleaning due to a loss of reflectance (e.g. a 30% loss of reflectance is quoted as a trigger for repainting). A review in ExternE (Friedrich and Bickel, 2001) looked at the functions relating to an exponential and a square root model, and reviewed critical levels and repair action. The most promising function was that of Pio *et al.* (1998): however, this function has proved difficult to implement in practice.

As a result, a simplified approach is often used that quantifies soiling damage based on cleaning costs (in the absence of WTP data). However, as this approach does not include amenity costs, it is therefore clear that cleaning cost estimates will be lower than total damage costs resulting from the soiling of buildings. There is one study that has incorporated an amenity component, described below. This provides a function which links population weighted particle concentrations to cleaning and amenity costs.

¹⁴ Jeanrenaud *et al* (1993) looked at the social costs of transport in Switzerland. Although this study adopted a top-down methodology, it did provide estimates for window cleaning costs. The study considered commercial buildings, assuming an average unitary cost of window cleaning of 6SF/m². The results indicate that window cleaning costs could be significant.

Rabl et al (1998) looked at total soiling costs (i.e. the sum of repair cost and amenity loss). The study showed that for a typical situation where the damage is repaired by cleaning, the amenity loss was equal to the cleaning cost (for zero discount rate); thus the total damage costs are twice the cleaning costs. The study recommended the following function:

$$S_i = a * P_i * \Delta TSP_i \quad (\text{where } a = b * 2)$$

S_i = Annual soiling damage at receptor location i .

P_i = Number of people in location i .

ΔTSP_i = Change in annual average TSP (Total Suspended Particles) $\mu\text{g}/\text{m}^3$.

a = WTP per person per year to avoid soiling damage of $1 \mu\text{g}/\text{m}^3$ particles.

b = Cleaning costs per person per year from a concentration of $1 \mu\text{g}/\text{m}^3$ of TSP.

This function allows a site-specific assessment, linking reductions in particle concentrations with population. A value of €0.5 for cleaning costs (b) has been used previously, based on Parisian data. In applying this function, a number of considerations are important:

- PM_{10} may not be the most relevant pollution metric for analysis. Instead black smoke or TSP (total suspended particulates) are better metrics for assessing damages.
- Knowledge of the characteristics of different types of particulates suggests that only primary particles have soiling effects. We assume that that secondary particles formed from SO_2 (e.g. sulphate aerosol and ammonium sulphate) and from nitrates (e.g. ammonium nitrate and nitrate aerosol) are very different in nature (they do not contain PEC) and do not lead to a loss of reflectance. Note, for national or city wide measurement data, the use of measured PM_{10} would therefore need adjustment for the proportion of primary and secondary particulates in the original air pollution mixture.
- There is also a question of a threshold for soiling. The loss of reflectance needed to trigger action, i.e. cleaning, will only occur when there is a certain build-up of particles. For this reason, observation shows that soiling is primarily associated with urban emissions of particles – there is no rural effect from low levels of building exposure. It is even likely that the effect is constrained to certain road types, notably street canyons, where buildings are extremely close to the roadside.

Soiling damage was not quantified in CAFE for two reasons:

- Pollution data were not easily available at an appropriate resolution
- A trial analysis showed that associated damage was small relative to other impacts

This decision was not revisited for EC4MACS. However, it may well be appropriate to include soiling if modelling were applied at a city-level, particularly for a scenario that considered significant reductions in primary particles.

7. Ecosystems

7.1. Types of impact

Pan-European action on air quality goes back to the 1970s following the surprise finding that soils in remote areas in Scandinavia were acidifying. The only explanation for these observations was that the problem was caused by long range transport and deposition of acidic pollutants generated in other parts of Europe.

The clearest effect of this long range deposition was probably the loss of salmon and trout from a large number of acidified rivers and streams in northern Europe.

Observations in the Black Triangle between Poland, and the former Czechoslovakia and German Democratic Republic focused concerns about air pollution's links to forest decline which had been seen previously in the English Pennines around industrial cities and in the Ruhr in Germany. Forest decline of this type was clearly linked to emissions of SO₂, and was severe in its impact, though restricted in range to a region of very high SO₂ emission, as opposed to an effect of long-term and long-range deposition. In locations more remote from major sources of emissions unusual symptoms were becoming evident on silver fir, Norway spruce, Scots pine, beech and other species. Declines were also noted in North America, raising concerns about the role of ozone in European forest decline.

A large research effort was mounted to investigate the linkage of air pollution to these forest declines, including initiation of the pan-European forest health surveys. Together with work on other semi-natural terrestrial ecosystems and on freshwater ecosystems this research provided a basis for defining critical loads¹⁵ for both acidification and eutrophication from airborne sources of pollution. However, definition of the precise role of air pollution in the observed forest declines remained elusive, as understanding of the various other stresses faced by forests increased.

Ecological sensitivity to air pollution is seen as being greatest in (semi-) natural vegetation, then forests, and least in crops. This ranking reflects many things, not least the level of human influence over genetic selection and also over processes affecting the soil and growth.

The effects of acidification and eutrophication can be expressed in general terms as causing ecological changes, which may be subtle (as in changes in the relative abundance of species in a particular ecosystem) or obvious (as in the elimination of salmon and trout from rivers and streams). Whilst air pollution should be seen as only one of a number of stresses that affect ecosystems in Europe, a reduction in pollutant emissions will reduce stress on ecosystems, which will assist in halting biodiversity decline. ICP/MM (2004) reports on numerous studies showing negative impacts in the performance of species in natural ecosystems. These impacts (e.g. a reduction in shoot growth or the amount of seed produced) should not of course be seen in isolation, as they affect the ability of species to maintain their status in ecosystems.

¹⁵ Critical loads are defined as a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to current knowledge. The sensitivity of different areas is variable, reflecting local conditions such as the availability of base cations in the soil that can neutralise acid inputs.

Risk is not spread evenly across Europe. For acidification the most sensitive areas tend to be those that receive high rainfall (through its effect on deposition rate) and where the bedrock (e.g. granite) weathers slowly and hence releases neutralising base cations at a rate that may be insufficient to match acidifying inputs. For eutrophication the most sensitive areas will again be those subject to high rainfall, but also areas that are naturally low in nitrogen. Ecosystems in these locations and the organisms that they contain have evolved specifically to cope with limited nutrient availability. Additional nutrient inputs enable other species (for example, some of the common grasses) to access areas in which they would previously have struggled to survive, or to grow much more strongly than they were previously able to, out-competing other less aggressive species, leading to changes in biodiversity.

Effects of ozone on natural ecosystems have also been noted (see ICP/MM 2004 for review material in the mapping manual).

7.2. Stock at risk data

Stock at risk data for ecosystem impacts has been collated over a period of many years through the Coordination Center for Effects in the Netherlands. This is already linked into the GAINS database, and no additional information will be required.

7.3. Exposure-response functions

Using methods agreed with the CCE (and hence already subject to extensive discussion with ecological experts across Europe) GAINS quantifies critical loads exceedance for acidification and eutrophication in terms of:

- Area in each country where ecosystems are exceeded;
- Accumulated exceedance of critical loads;
- Damage and recovery delay times.

Guidance from ICP/MM provides a consensus on characterising exceedance.

Ideally it would be possible to go beyond this, to describe impacts on different types of ecosystems in terms, for example, of effects on the abundance of different species across Europe, and recovery of ecosystems once critical loads are not exceeded. Further to this, consideration should be given to effects on ecosystem services, identifying a clear rationale for why people value ecosystems. It should be noted that this approach applies not only to productive functions (e.g. production of timber or wild edible mushrooms) but also to regulatory functions (water cycle, carbon cycle, etc.), cultural functions (including non-use values), etc.

However, whilst information from the literature provides insight on the types of effect that may be anticipated, there is a lack of information at the present time for going beyond this. Some case studies have been carried out (e.g. Hornung et al's, 1995 study of trout densities in some Welsh streams, following from Ormerod, 1990), but these tend to require further validation and cannot be extrapolated to the general European situation. The major problems for quantification lie in the need to account for effects over very prolonged timescales and the variability of conditions pertaining to climate, soil, the species that are present, ecological structure, human pressures and so on. Work in this field is continuing, for example through the EC funded ECLAIRE Project. Following the completion of EC4MACS this area of research will therefore continue, and the modelling framework extended if possible.

7.4. Valuation of ecosystem damages

The information provided in this section was written in the early phase of EC4MACS. Since then there has been significant new work published in relation to the ecosystem services approach. However, that is being reviewed elsewhere with a view to integration with the ALPHA-Riskpoll modelling framework (ECLAIRE project). Much of the following text, however, remains relevant.

7.4.1 Background and review

The monetary valuation of ecosystem benefits has until now not been included in the assessment of policy options. A major review of the literature on the valuation of ecosystem damages, acid rain, ozone, nitrogen and biodiversity was carried out by ECOLAS under an EC contract (06/11867/SV) to indicate the trajectory to be followed in order to enable the monetary assessment of ecosystem benefits of air pollution abatement policies. On the one hand this concerns the methodology to arrive at a European wide monetary assessment of ecosystem benefits of air pollution abatement. On the other hand this comes down to identifying the actions needed to further develop the methodological framework.

The concepts of critical loads and levels have been used for the development of air pollution policies in the European Union and in the framework of the Convention of Long-range Transboundary Air Pollution. However these concepts provide no information on the degree of damage to the ecosystem. Consequently they cannot serve as a direct input for the monetary assessment of ecosystem benefits (beyond identifying that areas are ‘at risk’).

There have been some studies on the monetary valuation of ecosystem benefits of air pollution abatement. Most of these were carried out in North America and Scandinavia. Also in the Netherlands the issue has received some attention and efforts were focused on the assessment of the welfare implications of acidification. With the exception of the efforts by the USEPA, in the framework of the evaluation of the Clean Air Act, there have been no coordinated initiatives. The existing valuation studies have little relevance to a comprehensive European wide assessment. The same goes for the elicitation of provisional monetary benefit estimates. The reasons for this are the following:

- The number of studies is limited. The coverage of the area, ecosystem types and number of ecosystem services potentially benefiting is limited.
- Very often the studies have been carried out because of the interest in certain monetary valuation methods. The scientific underpinning of the ecological aspects was therefore often of minor importance.
- Many dose-effect relations are quite uncertain.
- Many studies are quite old.
- The studies that used stated preference methods to elicit peoples’ willingness to pay for a given improvement in use values may also have elicited non-use values.

The methodology for the monetary assessment of ecosystem benefits of air pollution abatement can be broken down into three major phases. The first phase (exposure assessment) implies the determination of the relevant abatement scenarios and the resulting changes in ecosystem exposure to air pollution. Doing so involves the identification of those ecosystem areas that are meaningfully affected by the action.

The second phase (ecological response assessment) then involves the establishment of the appropriate linkages between the changes in ecosystems exposure to air pollution and the resulting effects.

The third phase (economic valuation) is about determining to what extent the quality and/or quantity of the ecosystem services benefiting from air pollution abatement changes given the effects on ecosystems. The (relevant) changes then need to be monetised. This can be done by using observed market data, performing a number of well-chosen valuation studies or transferring values from other valuation studies.

The ECOLAS report made the following recommendations:

1. In order to arrive at a European wide monetary assessment of the ecosystem benefits of air pollution a number of well-chosen original valuation studies will need to be carried out. The determination of the number, the ecosystem types and services covered, geographical delineation, etc. has to be underpinned by a considered strategy. Given the obvious resources constraints, this strategy should make optimal use of the possibilities of benefits transfer.
2. As the monetary valuation of ecosystem benefits of air pollution abatement is complicated by various problems it is impossible to attempt to overcome all of them at the same time. The trajectory for the future will therefore be one of stepwise improvements. The challenge of arriving at monetary estimates of ecosystem benefits of air pollution abatement requires on the one hand the well-considered deployment of research efforts and resources and on the other hand the search for a wide consensus about the methodology among scientists as well as decision makers.
3. In the light of this, it is advisable to have only a few ecosystem services covered by the assessment in the beginning. The criteria for selecting these service flows are quite straightforward. On the one hand there is the relevance of the change in the ecosystem service flow, and the associated welfare impact, resulting from air pollution abatement. On the other hand there are the feasibility, consensus and uncertainty of assessing the effect.
4. The different phases, and the components of which they are made up, in the methodology for the monetary assessment of the ecosystem benefits of air pollution abatement, linking emissions to welfare, need to be geared to one another. This is required to allow a sound and defensible estimation of the welfare change resulting from an action. For this the coordination between the exposure assessment, the ecological response assessment as well as the economic valuation have to be further stimulated.

7.4.2 Main valuation studies to date and further work needed

Notwithstanding the conclusions of the ECOLAS review it is worth looking at the literature on valuation of ecosystems and the damages from air pollution more closely. Earlier reviews concluded that, based on studies to the early part of this decade, the information was not good enough to form the basis of region-wide estimates of the damages to ecosystems (MacMillan et al, 2001; UNECE, 2003; Holland et al, 2005a). In the framework of the CAFE programme, Holland et al (2005a) concluded: “Although the literature in this area is growing, it is not currently adequate for a European wide appraisal such as this. Earlier studies tended to take a very simplistic perspective of impacts on ecosystems rendering them unsuitable for use in a policy context.”

In this section we look at areas where there are some positive results – i.e. where credible estimates have been made and where, with some work they could be built on to obtain estimates of damages in a Europe wide context. The areas covered are:

- Marine Ecosystems
- Forests
- Freshwater
- Complex of different ecosystems
- Cost savings

Marine Ecosystems

Marine ecosystems are affected by eutrophication and the problem is particularly serious in the Baltic Sea. The main benefits that have been estimated are recreational, relating WTP for areas that do not suffer from limited secchi depth and other impacts of eutrophication (Soutukovra, 2005; Gren et al. 1997; Standstrom, 1996). Links to nitrogen deposition, while complex are possible and, based on existing studies, it should be possible to estimate the costs of emissions from air on recreational values via eutrophication. The work is on the modelling side, linking air emissions to levels of eutrophication rather than on the valuation side, where marginal changes in levels of eutrophication have been estimated. Of course the consequences will vary across the marine water bodies and this has to be taken into account. But, in our view, this exercise is feasible.

Forests

The understanding of the relationships between the condition of forest ecosystems and the impacts of air pollution is still incomplete. The picture of air pollution effects on forests is a rather complex and subject to spatial variation. In many instances, air pollution is just one factor impacting on forest health. The focus of existing valuation efforts in forest ecosystems has mainly been on assessing the effects on trees.

A recent study by Karlsson et al (2005) represents a first attempt to assess the economic impacts of ozone on forest production in Europe. This was done for the Estate Östads Säteri in south-western Sweden. Harvests were estimated to be reduced by 1.8% because of ozone damage. The economic return, defined here as the difference between revenues and costs from harvests, was reduced by 2.6%. The greatest uncertainty in the estimates of ozone impacts on forest production is the up-scaling of ozone effects on growth, measured on young trees under experimental conditions, to mature trees under field conditions. The results arrived at for the Estate Östads Säteri are extrapolated to the whole of Sweden and all EU Member States. The authors state that this does not lead to consistent outcomes since the effects of ozone may differ due to different exposure levels. However, doing so provides an idea of the potential economic effects of ground-level ozone on forest production on a wider scale. The total loss to Sweden would be in the range of €56 million. For Europe this would result in a total loss of about €316 million (Karlsson et al, 2005).

To be useful to policy decisions, this study would need to be extended to look at damages from changes in ozone concentrations.

For acidification impacts on forests there are some older papers available (e.g. Gregory et al, 1996) though the methods used there are not valid, certainly at a pan-European scale. A problem for such studies was the lack of a single unifying damage mechanism, as many factors interacted to cause forest decline, and the strength of these factors was variable across Europe. So, in some areas, forest damage was clearly attributable to high emissions of acid

gases, whilst in others problems were linked to nutrient imbalances and in others again, to pest outbreaks or to several factors acting simultaneously. Many of these factors could be linked to air pollution, though not all.

As the ECOLAS report notes there are some important conclusions that can be drawn on the (monetary) assessment of the impacts on forest ecosystems. First of all, there is still considerable scientific uncertainty surrounding the link between air pollution and timber production. The dose-response relationships are not well-established. To an important extent, this uncertainty is due to fact that the interrelations with climate change effects as well as other natural stress factors makes it harder to provide evidence. Dose-response relations of both acidification and ground-level-ozone on forests have mainly been assessed under laboratory conditions using young trees. Second, it is also worth noting that the non-timber effects of air pollution on forests are barely addressed. Ozone for example may also have important aesthetic effects on forests.

In view of the above, and accepting that there will be gaps, further development of pan-European assessment of the effects of air pollution on forests is a feasible and worthwhile exercise.

Freshwater

The main effects here are acidification of lakes and streams and toxic impacts on fish from metals released into the water as a result of acid rain (e.g. aluminium). Freshwater ecosystems are also exposed to eutrophication, although the impact from air emissions on freshwater eutrophication is probably small relative to other sources. From a policy perspective it would seem strange to prioritise air emissions over agricultural emissions and wastewater discharges to limit freshwater eutrophication.

The main valuation studies of interest to us are those by Navrud (2002) and USEPA (1999). The USEPA study looked at the benefits and costs of the Clean Air Amendments of 1990, and as part of those the annual economic impact of acidification of freshwater fisheries was assessed. The study focused on the lakes of the Adirondacks national park. The annual benefits accruing to the New York State residents were calculated to be in the range of \$12-49 million using an effects threshold of pH 5.0 and in the range of \$82-88 million for an effects threshold of pH 5.4. The obvious critique on the approach of this study is that it assumes a simplistic binary damage response function (fish/no fish) and thus does not allow for intermediary values.

Navrud (2002) assessed the willingness to pay of the Norwegian population for increased fish stocks, resulting from reduced exceedance of sulphur critical loads. The approach used in this study goes from emissions to benefits by (1) linking changes in emissions to critical load exceedance, (2) applying dose-response relationships for exceedance of critical loads and damage to fish and (3) applying economic valuation methods to assess the impact of fish damages on human welfare. The valuation exercise consisted of a national wide contingent valuation study to estimate the monetary value of the increased number of lakes with undamaged fish stocks from acidification. The estimated yearly benefit to all Norwegian households would be in the range of €80 to €134 million.

In principle the valuation of acidification on fish stock and thereby on recreational and commercial values is possible, and the above studies attest to that fact. More work is needed to develop the model used by Navrud to other freshwater systems and to combine it with

some more valuation work. Based on that, estimates of the damages from emissions in different parts of Europe should be possible.

Complex Systems

Studies exist that value whole changes in ecosystems. The numbers are more difficult to use because they relate to changes associated with whole programs (e.g. the Clean Air Amendments in the USA) and are not attributed to programme components, which is what is needed for the values to be used in a wider range of applications. In addition going from total values to marginal values is problematic, especially if non-linear effects (e.g. via thresholds) are present. This approach is therefore not recommended for further examination in EC4MACS.

Cost Savings

In some cases reductions in emissions reduce costs of economic activities, such as treatment of drinking water, liming of lakes etc. Although these are not damage estimates they can be useful if one could reasonably assume that such actions are justified and that if they were not undertaken, the damages would be greater than the costs. Consideration should be given to the use of these estimates. Some caution should be exercised to ensure that results are not presented in a way that implies they give a more complete picture of the change in ecosystem related damage than is really the case.

7.4.3 Conclusions

This survey shows that there needs to be more work done before a satisfactory monetary valuation of the damages from air pollution to ecosystems can be performed. This is being taken forward through ECLAIRE and other projects.

8. Other Impacts

8.1. Social effects

There are several issues to take account of here:

1. Variation of exposure to air pollution amongst communities who rate poorly on social deprivation indices (King and Stedman, 2000; Pye, 2001).
2. Variation in susceptibility of different groups to health impacts. Krewski et al (2000); Hoek et al (2002) and Pope et al (2002) have observed links between air pollution impacts and low educational attainment. Abbey et al (1999) and Brunekreef (1999) found higher susceptibility to air pollution effects amongst subjects with poorer nutrition.
3. The ability of individuals to mitigate impacts, for example through the purchase of medicines and access to good quality health care or routine maintenance of buildings.

It may be that other issues should be added to the list provided here – we are unaware of previous reviews in this area, though there is work that deals with some of the elements that are identified.

The assessment of links between air pollution impacts and social deprivation under EC4MACS will be limited quantitatively because of a lack of data. There are several reasons for this:

1. Systems for reporting social deprivation will vary from country to country;
2. There is a lack of information on:
 - a. How the various elements that make up indices of social deprivation are linked to air quality;
 - b. How the health of groups that differ in social deprivation responds to changes in air quality (as opposed to the population as a whole, though see Figure 4 above, which indicates that effects of air pollution are greatest on those with reduced life expectancy);
 - c. Response to air pollution effects amongst different groups in the community.
3. The analysis would require data at a much higher resolution than is currently available;
4. Analysis at the required resolution on a European scale would be extremely time consuming, even if data were available.

Requests have periodically been made for more data on links between social deprivation and air pollution but without response.

In describing impacts on equality it is important not to be too alarmist about the role of air pollution in determining social disadvantage as other factors are far more important in this respect (education, structure of welfare systems, etc.). However, it is clear that there are reasons for some concern.

Some caution is also needed with respect to possible response to inequalities. Even if links were strong it would not necessarily be appropriate to target air quality improvement measures specifically on the socially disadvantaged. This could lead to the selection of generally less cost-effective measures, with the consequence that the overall change in

population exposure, including exposure of the more vulnerable groups, is less than that achievable without any attempt to account for the more disadvantaged.

8.2. Visibility

Analysis in the USA has concluded that reduced visibility is a significant impact of air pollution (USEPA, 1997). The word ‘visibility’ in this context relates to a reduction in visual range caused by the presence of air pollutants in the atmosphere. The problem is associated largely with particles and NO₂. At pollutant levels typical of Europe and North America this can lead to impacts on amenity in terms of reduced enjoyment of landscapes.

In Europe, the association of air pollution with reduced visibility has received very little attention. There are several possible reasons for this. Perhaps the most important is that there have been significant improvements in visibility already across much of Europe. In London, for example, sunshine hours in January doubled in the 20 years that followed the adoption of the UK’s first Clean Air Act in 1956 (Chandler and Gregory, 1976), and air quality in the city has continued to improve since.

Following review of quantification methods for visibility impacts (see Appendix 10 of Holland et al, 2005a) it was concluded that there was an inadequate base of European data on which to base a credible assessment. The peer reviewers supported this position. Amongst the stakeholders who commented on drafts of the methodology report some supported, none actively disagreed. The impact does not therefore form part of the core analysis for the EC4MACS benefits assessment.

8.3. Climate change

The effects of climate change are not assessed within the EC4MACS benefits modelling. However, many of the measures that could be introduced to further control regional air pollution will have an effect on greenhouse gas emissions and vice versa:

- Reductions in emissions of the main greenhouse gases will arise (for example) through falls in energy demand via measures that improve energy efficiency or promote less polluting modes of transport;
- Increases in greenhouse gas emissions may also arise (for example) through the use of end-of-pipe pollution controls that require some energy input.

There are two approaches valid to valuing the change in emissions under slightly different situations:

- When considering reductions in greenhouse gases that go beyond existing agreements such as the Kyoto Protocol the damages to health, environment, society and economy should be applied.
- When considering changes in greenhouse gas emissions that are within the limits prescribed by existing GHG control agreements the relevant value is the marginal cost of GHG abatement. It would be assumed that the change in emissions simply made it easier or harder to meet the necessary limits, and that overall climate impacts to health, the environment, society and the economy would not be changed.

There have been several studies and reviews (e.g. the Stern Review and the EC funded ClimateCost study¹⁶) that have estimated the costs of climate change damages per tonne CO₂-eq emitted.

Turning to estimates of the marginal costs of control, Holland et al (2005a) considered the following to be most appropriate although these values were never applied in the CAFE-CBA work. Analysis by European Climate Change Programme (ECCP) working groups in June 2001 identified 42 possible measures, which were estimated to reduce emissions by 664-765 MtCO₂ for a cost of less than 20€/tonne CO₂eq. In a study that underpinned the ECCP¹⁷ process it was found that the EU15 would be able to comply with its –8% greenhouse gas reduction target at a marginal cost of €20/tCO₂ eq. Given that the enlargement of the EU has given a possibility of lowering compliance costs, and given that through Joint Implementation and Clean Development Mechanism projects it is possible to reduce compliance costs, the Climate Change Unit of DG Environment advised the CAFE programme to apply the following compliance costs.

Table 16. Assumed compliance cost to reduce a tonne of CO₂ equivalent in 2010, 2015 and 2020

2010	2015	2020
€12 /tonne CO ₂ eq	€16 /tCO ₂ eq	€20 /t CO ₂ eq

These figures will vary over time, reflecting advances in technology and changes in society. When such estimates are integrated to analysis reference should be made to the position of the sponsoring body (e.g. government ministry, DG CLIMA, etc.) at that time.

8.4. Effects of emission control measures on emissions of other pollutants

The measures adopted for further control of air pollution in Europe may affect emissions of a wider range of pollutants than those considered core to EC4MACS and climate policy (see Section 8.3). These may include PAHs, heavy metals, dioxins, CO and benzene, depending on the abatement techniques adopted and sources addressed. Past CBA work on these pollutants under the Air Quality Framework Directive (AEA Technology, 1999; 2001; Entec, 2001) suggests that benefits of control are less significant than for the main CAFE pollutants. Detailed quantification does not, therefore, seem appropriate. However, the issue is being kept under review – for some sector-specific analysis it could be important.

8.5. Effects on groundwater quality and drinking water supply

Research in the Netherlands (van der Velde et al, 2004) has investigated the benefits arising from reduced acidification in terms of:

- Lower costs for treatment of groundwater;
- A longer life time for wells, pipelines and other hardware;
- Lower maintenance costs for wells and pipelines;

¹⁶ <http://www.climatecost.cc/>

¹⁷ Economic Evaluation of Sectoral Emission Reduction Objectives for Climate Change available at http://europa.eu.int/comm/environment/enveco/climate_change/sectoral_objectives.htm

- The benefits for the water treatment are negligible due to the low margins (WP/NP) in the groundwater.

In total a net present value of €45 million was estimated for the benefits to the Netherlands for the period 1990 to 2040. Benefits may increase significantly if the time frame were to be extended, though the total benefit appears too small to have major consequences in terms of the other effects considered in the EC4MACS benefits assessment.

The authors of the paper discuss possible benefits in the rest of Europe and conclude that they could be much higher than in the Netherlands. Reasons for this were that other countries make more use of vulnerable materials (e.g. cast iron) in wells and water delivery systems, and often make more use of resources where the travel time of water is short (e.g. surface waters).

9. Comparing costs and benefits, factoring in uncertainties

Methods for uncertainty assessment were discussed by Holland et al (2005b) and further developed and implemented in the various analyses performed to inform the development of the Thematic Strategy on Air Pollution and the revision of the National Emission Ceilings Directive (AEA Energy and Environment 2005a, b, c; 2006, 2008). Within the EC4MACS project the TUBA Framework (Treatment of Uncertainties in Benefits Assessments) has been formalised, to collate all available information on uncertainty. This is most useful when applied in CBA (rather than an isolated impact or benefit assessment), as the process of comparison gives context to the scale of uncertainty: If there is a substantial distance between estimates of cost and benefit (e.g. a factor 10) for a particular policy, uncertainties become irrelevant to the decision making process (at least from an analytical perspective). If the two are of a similar order of magnitude it is quite possible that uncertainties could affect the conclusions drawn from results.

9.1. Basic comparison of costs and benefits

Best estimates of costs generated by GAINS can be compared with the best estimates of benefits generated by ALPHA2/ALPHA-Riskpoll to provide a preliminary impression of the balance of costs and benefits. This should be done (to the extent possible) by considering marginal costs and marginal benefits. Within the modelling framework of EC4MACS a true marginal comparison is difficult, so instead a proxy is used – the difference between scenarios. The smaller the difference between the scenarios, the closer the analysis is to a true marginal assessment. The use of (near-) marginal CBA is often not apparent from final papers written to inform European air pollution policy as they tend to focus on whatever scenario has been recommended, and then describe the costs and benefits of that scenario relative to business as usual without reference to the way that costs and benefits relate to one another for intermediate positions. However, a larger number of scenarios will typically have been evaluated previously to facilitate a near-marginal comparison.

The most basic form of CBA test is simply to see whether costs exceed benefits or vice versa. If benefits are greater it is appropriate to quantify the ratio of benefits to costs to derive information on the effectiveness of expenditures.

It is also important to develop an understanding of the robustness of the relationship between the two sides of the CBA equation, and this part of the analysis needs to account for uncertainties in the assessments. The rest of this Chapter provides a methodology for assessing the effects of these uncertainties.

9.2. Types of uncertainty

9.2.1 Statistical uncertainties

Discussion of uncertainty often focuses purely on those aspects of analysis that can be quantified using statistical techniques. These techniques address uncertainty associated with the extraction of information from observations on a limited sample drawn from a population of people, crops, industrial plant, etc. They describe the behaviour of the sample (e.g., how it

responds to change in a variable such as increased air pollution) and show how reliable the conclusions drawn from use of the sample are as a representation of the behaviour of the total population. Key characteristics of a sample are average (also referred to as ‘mean’) or median values and the spread of values around them. Spread is typically characterised as the standard deviation and the range within which 90, 95 or 99% of observations are likely to occur.

Whilst statistical analysis provides a benchmark for uncertainty assessment it is important to recognise that a variety of uncertainties cannot be described using standard statistical techniques:

- Omission of impacts from the benefits analysis.
- Existence of alternative views on methodology amongst experts (e.g. in relation to mortality valuation).
- Transfer of data on exposure-response, valuation, etc. from one situation to another.

Although a statistical treatment of these uncertainties is not possible, it is still necessary to account for them in the analysis in some way if they seem likely to have a significant effect on the balance of costs and benefits. This can be done using the other techniques described in this report, bias analysis and sensitivity analysis.

The further the analysis proceeds through the chain from release to exposure to impact assessment to valuation, the greater the uncertainty in the final estimate (simply because more parameters, each bringing their own level of uncertainty to the analysis, are introduced). On this basis, we can have the highest confidence in concentration data, followed by (in order) total population exposure, exposure of specific groups within the population, impact results, and finally monetised estimates of damage. That said, it is important to recognise that the growth of uncertainties down the chain of analysis is not exponential, as there is the likelihood of different uncertainties operating against each other to cancel to a greater or lesser extent, assuming of course that there is not a systematic and consistent bias in all inputs.

9.2.2 Biases

Biases reflect limitations in the tools available for quantification of (in this case) the costs and benefits of pollution control. They are issues for which quantification and associated assessment of uncertainty in a sufficiently detailed manner for inclusion in the analysis is not possible. They need to be brought into the assessment in some way because many of them have the potential to influence results significantly (e.g. the omission of secondary organic aerosols from the dispersion modelling, of abatement options from GAINS, and of ecosystem damage from the benefits assessment). In many cases the direction of the bias on the balance of costs and benefits is obvious. In a few cases, however, it is not.

The treatment of biases proceeds through the following stages:

- Identification of biases
- Assessment of the direction of bias
- Assessment of the potential effect of biases on the cost-benefit balance
- Interpretation of the overall effect of the biases identified.

9.2.3 Sensitivities

There are several methods available that come under the general title of sensitivity analysis:

- Observation of the effect on outputs of a systematic stepwise change in one or more variable(s). This could, for example, involve assessment of the effect of a series of incremental changes of 5% or 10% around the core estimate for a specific variable.
- Use of alternate estimates for a specific parameter based on different methodologies. Examples include:
 - Monetisation of mortality impacts using VOLY (value of a life year) and VSL (value of statistical life) based methods.
 - Use of European average or country specific valuations.
 - Use of different approaches to discounting.
- Division of impacts into confidence bands, to differentiate between those effects that can be assessed with greatest confidence and those that can be quantified with less confidence.

Cost-benefit analysis of the National Emission Ceilings and Ozone Directives and the Gothenburg Protocol (AEA Technology, 1998; Holland et al, 1999) used two forms of sensitivity analysis. First, it grouped quantifiable benefits into five confidence bands, demonstrating the confidence of stakeholders and analysts in the quantification of each impact. Those effects for which quantification was considered most robust were put into confidence band 1, whilst those for which quantification was considered least robust were placed in confidence band 5. A stepwise comparison was then made with costs. If the benefits from the impacts in confidence band 1 outweighed abatement costs for any country, that country would have great confidence that overall, benefits would outweigh costs. If all five confidence bands were required, confidence that benefits would outweigh costs would be lower (acknowledging that some important impacts were left out of the quantification altogether, as now). A weakness of the approach is that there is subjectivity in defining which effects can be quantified with greatest confidence. It was noted that few stakeholders felt able to respond to a questionnaire distributed at the time to solicit opinion on how impacts should be ranked. Of those that did, many expressed the view that they could only comment on the impacts (e.g. health effects) with which they were most familiar. Although the approach was clearly not perfect, a number of stakeholders found it useful.

The second method used in the earlier CBAs was the separate investigation of the effect of individual sensitivities, with particular attention given to mortality valuation using the value of statistical life (VSL) and value of life year (VOLY) approaches, and the use of European average and country specific data for valuation. Consideration of the VOLY/VSL sensitivity is continued to this day.

9.2.4 Model validation and quality control

There is a risk of error in any analysis during model construction, the handling of data and processing and handling of results. The complexity and multi-disciplinary nature of the CAFE analysis raises the potential for such error. The developers of the EMEP and GAINS models have their own protocols for dealing with the issue of model validation and quality control. For the benefits analysis component of the CBA, the approach for dealing with uncertainty due to model validation and quality control has been as follows:

- Two modelling tools are available, ALPHA (Ricardo-AEA) and ALPHA-Riskpoll (EMRC), developed separately and hence permitting results to be compared for each endpoint.

- A series of marginal damage estimates per tonne pollutant emission have been generated by the project team. These can be used to check results of a full scenario analysis.
- The health functions provided in the methodology report require some computation before integration with the model (e.g. in converting odds ratios to change in incidence per unit pollution). All functions were checked independently of the main authors at IOM by EMRC during the writing of Volume 2 of the CAFE-CBA Methodology report (Hurley et al, 2005).
- Results have been compared against background rates, crop yield, etc., to assess whether or not they are plausible.
- Whilst direct validation is not possible, consideration has been given during the development of the benefit assessment methodology to information that shows impacts to be real. A good example would be the various ‘intervention studies’ that show significant changes in mortality and morbidity rates following a large stepwise change in emissions.

9.3. Statistical uncertainties

The approach used here for statistical analysis is based on the use of the @RISK model. @RISK permits investigation of statistical uncertainties through the definition of probability distributions for key parameters in terms of mean values and the spread of values around them, and subsequent sampling across these distributions.

The first stage in the analysis is definition of the scope of the model to be used for quantifying uncertainty. Based on the results from CAFE and the NECD revision, it is appropriate to focus on health impacts, as they provide the largest monetised air pollution damage for the EC4MACS analysis. The next step is to identify the different stages of the analysis, and the areas where quantifiable uncertainties are likely to be most significant. Probability distributions (illustrated in Figure 7) are then defined for each parameter of interest, drawing particularly on data given by Hurley et al (2005).

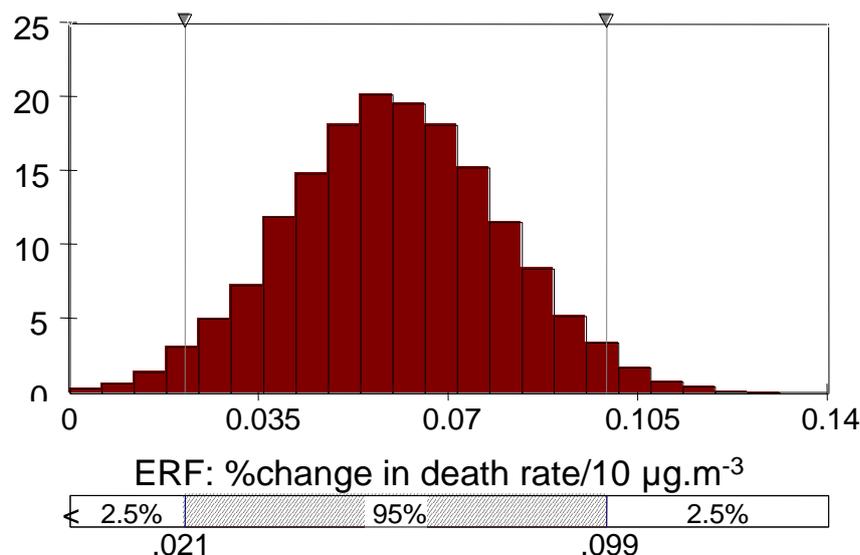


Figure 7. Normal distribution for exposure response function for chronic mortality effects of PM exposure.

Table 17 and Table 18 provide data on the ranges and best estimates entered into the @RISK model as used for analysis in CAFE-CBA. These have been retained in EC4MACS. From comparison of best estimates and standard errors, it may at first sight appear that uncertainty in valuation of mortality is underestimated compared to uncertainty in valuation of morbidity. However, a large part of the uncertainty in mortality valuation is accounted for by the reporting of separate results based on the median and mean estimates of the VOLY and VSL. Against this background, the use of broader spreads than those recommended here around the separate mortality estimates would double count uncertainties.

Table 17. Best estimates and ranges used for incidence data, exposure response functions and valuation data in the analysis of statistical uncertainties in the health impact assessment for ozone effects, based on information presented by Hurley et al (2005).

Annual incidence rate: distribution – triangular	+/-	Best estimate
Mortality rate (deaths per head of population)	5%	0.011
Respiratory hospital admissions, >64 years (cases/100,000 population)	20%	2,496
Minor restricted activity days (per person)	40%	7.8
Adult use of respiratory medication (days per person)	40%	0.045
Respiratory symptoms, adults (dummy variable)	40%	1
Response function: distribution - normal	Std deviation	Best estimate
Acute mortality (% change in mortality rate per 10 $\mu\text{g.m}^{-3}$)	0.075%	0.30%
Respiratory hospital admissions (%change in incidence/10 $\mu\text{g.m}^{-3}$ O ₃)	0.35%	0.50%
Minor restricted activity days, population 18-64 (%change in incidence/10 $\mu\text{g.m}^{-3}$ O ₃)	0.45%	1.48%
Respiratory medication use (days /10 $\mu\text{g.m}^{-3}$ O ₃ /1000 adults aged 20+)	456	730
Minor restricted activity days, (%change in incidence for population aged >64 /10 $\mu\text{g.m}^{-3}$ O ₃)	0.45%	1.48%
Respiratory symptoms (symptom days/1000 adults/10 $\mu\text{g.m}^{-3}$ O ₃)	175	343
Valuation (all units - €/case): distribution – normal	Standard error	Best estimate
Acute mortality (VOLY, mean) (€/case) ¹	14,600	120,000
Acute mortality (VOLY, median) (€/case) ¹	3,700	52,000
Respiratory hospital admissions (€/event)	670	2,000
Minor restricted activity days, (€/day)	13	38
Respiratory symptoms in adults (€/day)	13	38
Respiratory medication use by adults (€/day)	0.33	1

Notes: 1) To be revised

Table 18. Best estimates and ranges used for incidence data in the analysis of statistical uncertainties in the health impact assessment for PM effects. Data are based on information presented by Hurley et al (2005).

Annual incidence rate: distribution - triangular	+/-	Best estimate
Mortality rate, >30 years	5%	1.61%
Infant mortality rate, ages 1 to 12 months	10%	0.19%
Chronic bronchitis, % of population aged >27 years affected	40%	0.38%
Respiratory hospital admissions (cases/100,000 population)	20%	617
Cardiac hospital admissions (cases/100,000 population)	20%	723
Restricted activity days (RADs, days / person)	40%	19
Use of respiratory medication by adults (% symptomatic adults)	40%	4.50%
Use of respiratory medication by children (% of children who are symptomatic)	40%	20%
Lower respiratory symptoms, adults (% of adults who are symptomatic)	40%	0.30
Lower respiratory symptoms, children (dummy variable)	40%	1
Consultations asthma (consultations / 1000 children)	40%	47.1
Consultations asthma (consultations / 1000 adults of working age)	40%	16.5
Consultations asthma (consultations / 1000 elderly)	40%	15.1
Consultations URS consultations / 1000 children)	40%	574
Consultations URS (consultations / 1000 adults of working age)	40%	180
Consultations URS (consultations / 1000 elderly)	40%	141
RADs, young + elderly (days/person)	40%	19
Response function: distribution – normal	Std deviation	Best estimate
Mortality (change in mortality risk / 10 $\mu\text{g.m}^{-3}$ PM_{10})	2%	6.00%
Mortality (years of life lost (YOLL)/ $\mu\text{g.m}^{-3}$)	11	65.1
Infant mortality (change in mortality risk / 10 $\mu\text{g.m}^{-3}$ PM_{10})	1.00%	4.00%
Chronic bronchitis, >27 years (%change in incidence/10 $\mu\text{g.m}^{-3}$ PM_{10})	3.70%	7.00%
Respiratory hospital admissions (%change in incidence/10 $\mu\text{g.m}^{-3}$ PM_{10})	0.26%	1.14%
Cardiac hospital admissions (%change in incidence/10 $\mu\text{g.m}^{-3}$ PM_{10})	0.15%	0.60%
Restricted activity days (%change in incidence/10 $\mu\text{g.m}^{-3}$ PM_{10})	0.03%	0.48%
Use of respiratory medication by adults (additional days of bronchodilator usage per 1000 symptomatic adults per 10 $\mu\text{g.m}^{-3}$)	900	908
Use of respiratory medication by children (additional days of bronchodilator usage per 1000 children per 10 $\mu\text{g.m}^{-3}$)	430	180
Lower respiratory symptoms (symptom days / symptomatic adult /10 $\mu\text{g.m}^{-3}$ PM_{10})	0.57	1.30
Lower respiratory symptoms (symptom days/child aged 5-14/10 $\mu\text{g.m}^{-3}$ PM_{10})	0.45	1.85
Consultations asthma (% increase in consultations amongst children/ 10 $\mu\text{g.m}^{-3}$ PM_{10})	1.25%	2.50%
Consultations asthma (% increase in consultations amongst working age adults / 10 $\mu\text{g.m}^{-3}$ PM_{10})	0.95%	3.10%
Consultations asthma (% increase in consultations amongst the elderly / 10 $\mu\text{g.m}^{-3}$ PM_{10})	2.30%	6.30%
Consultations URS (% increase in consultations amongst children / 10 $\mu\text{g.m}^{-3}$ PM_{10})	0.35%	0.70%
Consultations URS (% increase in consultations amongst the working age population / 10 $\mu\text{g.m}^{-3}$ PM_{10})	0.45%	1.80%
Consultations URS (% increase in consultations amongst the elderly/ 10 $\mu\text{g.m}^{-3}$ PM_{10})	0.80%	3.30%
RADs, young+elderly (% change in incidence/10 $\mu\text{g.m}^{-3}$ PM_{10})	0.03%	0.48%

Table 18 (continued).

Valuation (all units – €/case): distribution – normal	Standard error	Best estimate
Chronic mortality (VOLY, mean, €/life year) ¹	14,600	120,000
Chronic mortality (VOLY, median, €/life year) ¹	3,700	52,000
Chronic mortality (VSL, mean, €/death) ¹	235,000	2,000,000
Chronic mortality (VSL, median, €/death) ¹	74,000	980,000
Infant mortality (€/death) ¹	1,000,000	3,000,000
Chronic bronchitis, >27 years (€/case)	63,000	190,000
Respiratory hospital admissions (€/event)	670	2,000
Cardiac hospital admissions (€/event)	670	2,000
Restricted activity days, working age (€/day)	27	82
Lower respiratory symptoms, adults and children (€/day)	13	38
Consultations asthma, URS (€/event)	18	53
RADs, young, elderly (€/day)	23	69
Use of respiratory medication (€/day)	0.33	1

Notes: 1) To be revised

The distributions given in Table 17 and Table 18 can be brought together within @RISK across all effects to provide an overall estimate of the probability distribution around the best estimate of benefits for any scenario. This is illustrated for two of the CAFE scenarios (the ‘low’ ambition Scenario A [Table 19 and Figure 8], and MTFR, a scenario that assesses the Maximum Theoretically Feasible Reduction in emissions according to the measures included in GAINS [Table 20 and Figure 9]). The results shown in the Figures are expressed as net benefits – i.e. after subtraction of costs. The Figures and Tables include not only assessment of statistical uncertainties, but also assessment of the effect of sensitivity to the way that mortality is valued. Whilst this sensitivity will continue to be explored in EC4MACS, some simplification appears possible by moving to only one case each for VOLY and VSL instead of two. The Tables include quantification of the probability that benefits will exceed costs. For scenario A this is very high (>99% in all cases) whilst for MTFR it is low (a maximum of 33%).

9.4. Sensitivities

From consideration of the data used to quantify impacts, the following possible areas for sensitivity analysis are apparent:

- Valuation of mortality in terms of lives impacted or life years lost, using the median or mean value of statistical life (VSL) or value of a life year (VOLY).
- Inclusion of a 4% risk factor for chronic mortality effects, in addition to 6% (as recommended by WHO)
- Accounting for variation in the health risk associated with different types of particle.
- Use of a cut-point for the ozone-health analysis.
- Inter-annual variability in meteorology (in cases where this has not been accounted for explicitly in the modelling).
- Inclusion of some health endpoints which are supported by limited research.

The need to consider each sensitivity is scenario dependent: there is clearly no point in considering a sensitivity if there is no chance that it will affect the conclusions reached. To illustrate, the inclusion of additional health endpoints (final bullet point) is unnecessary if there is a high probability that benefits exceed costs already without them.

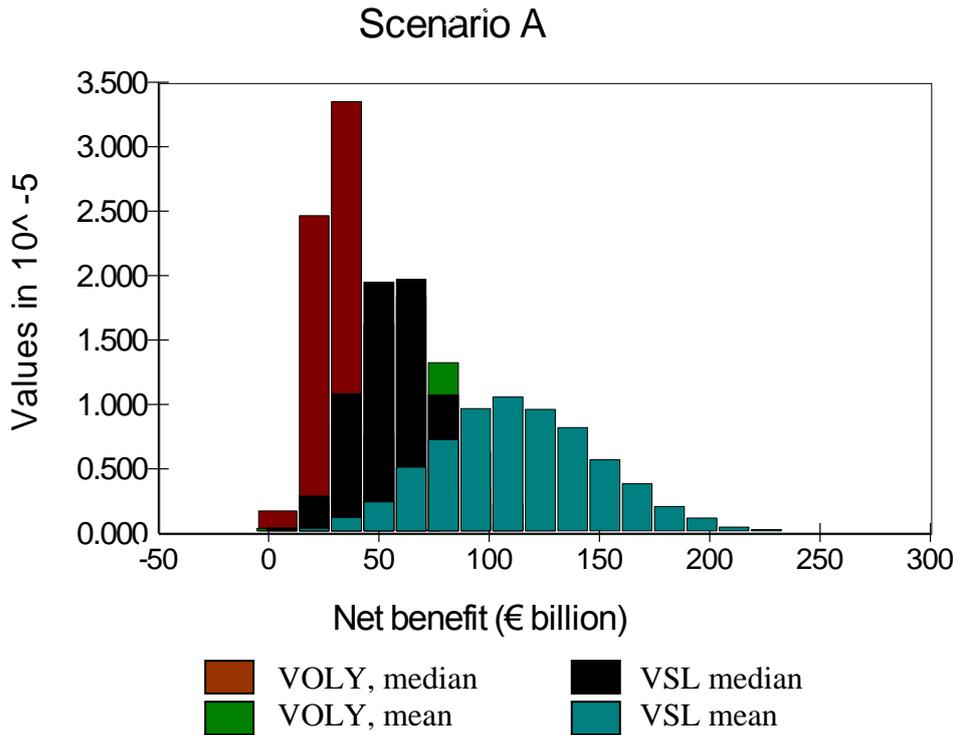


Figure 8. Probability distributions showing net benefit (benefit – cost) for proceeding from the baseline to Scenario A, with sensitivity to different approaches to mortality valuation also shown.

Table 19. Annual costs and benefits for the EU25 of proceeding from the baseline to Scenario A, and the probability that benefit will exceed cost.

	Cost (core estimate, € billion)	Benefit (core estimate, € billion)	Net benefit (core estimate, € billion)	Probability that benefit > cost
VOLY – median	5.9	37	31	>99%
VOLY – mean	5.9	70	64	>99%
VSL – median	5.9	64	58	>99%
VSL – mean	5.9	120	114	>99%

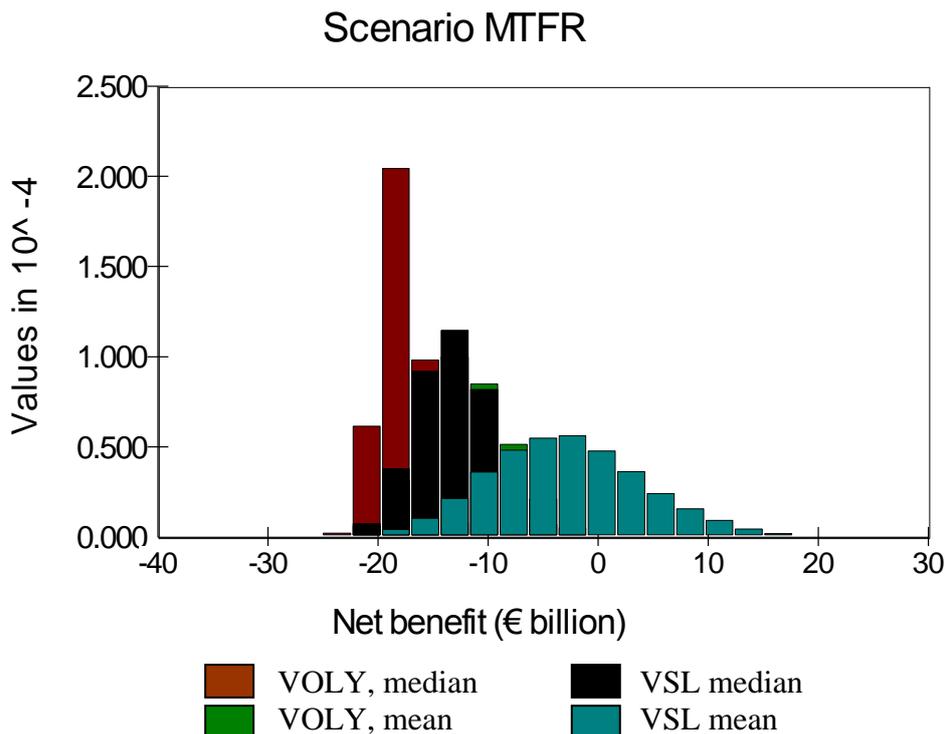


Figure 9. Probability distributions showing net benefit (benefit – cost) for proceeding from Scenario C to the MTFR scenario, with sensitivity to different approaches to mortality valuation also shown.

Table 20. Annual costs and benefits of proceeding from Scenario C to MTFR, and the probability that benefit will exceed cost.

	Cost (core estimate, € billion)	Benefit (core estimate, € billion)	Net benefit (core estimate, € billion)	Probability that benefit > cost
VOLY – median	25	6.9	-18	0%
VOLY – mean	25	13	-12	<1%
VSL – median	25	12	-13	<1%
VSL – mean	25	22	-2.9	33%

From a theoretical perspective it may not seem appropriate to treat uncertainty in cost estimates using sensitivity analysis. However, it was treated this way in CAFE because of a lack of information on the probability distribution for costs around the IIASA estimates. From the perspective of understanding the costs, specifically, this approach supplies no additional information. However, it is useful for comparing the robustness of the relationship between costs and benefits. The Monte Carlo analysis was run for the benefits assessment for a series of discrete cost estimates in the range of 50% to 120% of the GAINS forecast (see Figure 10, taking the example of Scenario C, considered ‘high’ ambition under CAFE). This example shows that if costs are significantly overestimated there is a high probability of achieving a net benefit irrespective of the approach taken to mortality valuation. So far relatively few cost-curve studies have so far attempted to generate a probability distribution (an exception being the work of Handley et al (2001) on the costs of abating non-agricultural ammonia emissions).

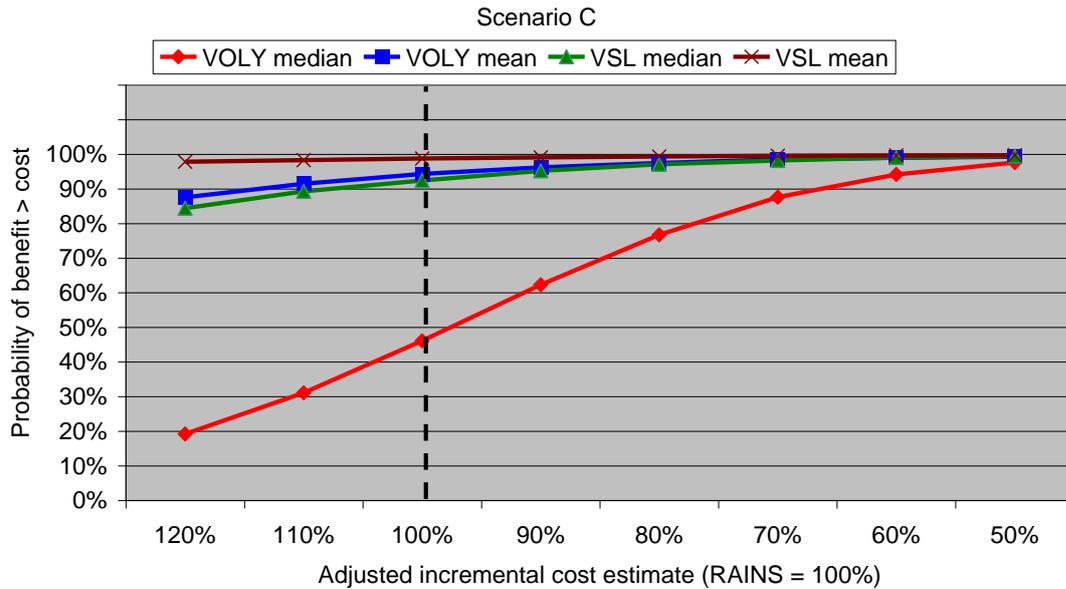


Figure 10. Sensitivity to uncertainty in incremental costs of pollution abatement of the probability of a net benefit in moving from Scenario B to Scenario C.

Two issues have not been considered here that some may have considered worth addressing:

- The use of a threshold for assessment of PM effects. This issue has been addressed in detail by WHO in their input to the CAFE process. In addition to the views expressed there, it is to be remembered that the EMEP and RAINS dispersion modelling excludes non-anthropogenic particles and secondary organic aerosols. On this basis the analysis already operates with an effective threshold of several $\mu\text{g.m}^{-3}$.
- The use of alternative factors for $\text{PM}_{2.5}$ to PM_{10} conversion for some of the morbidity functions. We do not consider that these would have a major effect on the analysis, particularly because the dominant PM effect (mortality) is characterised directly against $\text{PM}_{2.5}$.

It would of course be possible to add in further sensitivity analysis. However, there has to be a clear focus in uncertainty assessment that it is intended to improve understanding of results and their robustness. The addition of further sensitivities may simply serve to confuse.

9.5. Biases

When comparing biases and their effects on the balance of costs and benefits it is necessary to consider not only the benefits assessment based around ALPHA2, but also the EMEP dispersion modelling and the GAINS integrated assessment modelling.

9.5.1 Biases in the EMEP model

With respect to the two main pollutants considered in the EC4MACS benefits assessment, $\text{PM}_{2.5}$ and ozone, the 2004 review of EMEP concluded that:

- *The model, in the form presented, underestimated observed PM_{10} and $\text{PM}_{2.5}$ due to an incomplete description of relevant processes and emissions. It was, however, able to*

calculate the regional component of the main anthropogenic PM fractions (sulphate, nitrate, ammonium, some primary components) with enough accuracy to assess the outcome of different control measures. Note: For the final assessment under EC4MACS modelling included secondary organic aerosols. However, it is understood that there is still some ‘missing PM’ based on comparison of modelled and monitored levels.

- *The model shows an excellent level of performance for daily maximum ozone concentrations.*

These and other issues are considered in Table 21.

Table 21. Biases in the EMEP modelling. Ratings given in brackets are biases that are unlikely to have a significant effect on the outcome of the CAFE modelling.

Source of bias	Likely effect on benefit:cost ratio	Comment
Variability in meteorology from year to year	(+++/-)	The importance of this factor is dependent on how many years’ data have been used for the analysis.
Underestimation of suspended particle concentrations	---?	For the final assessment under EC4MACS modelling included secondary organic aerosols. It is understood that there is still some ‘missing PM’ based on comparison of modelled and monitored levels.
Lack of specific account of urban concentrations of: <ul style="list-style-type: none"> • PM_{2.5} 	0 (assuming CITYDELTA adjustment is correct)	Urban concentrations of PM are factored into the RAINS model using the results of the CITYDELTA Project. Ozone concentrations are generally depressed in urban areas as a result of high local NOx emissions.
<ul style="list-style-type: none"> • Ozone 	++	

9.5.2 Biases in the GAINS modelling

The peer review of the RAINS model (precursor of GAINS) carried out as part of the CAFE Process provides detailed consideration of both statistical uncertainties and biases (Swedish Environmental Research Institute, 2004). This has been used as the basis for the information presented in Table 22. Whilst the table has not been updated with specific reference to GAINS it is thought likely by the authors of this report that the overall picture is similar. Some of the biases identified in the review are not discussed in the table as they are not considered relevant here. For example, the omission of impacts of particles and ozone on morbidity is addressed through the wider quantification of benefits in the benefits assessment¹⁸. Nonetheless, the modelling is clearly subject to a significant number of possible biases.

¹⁸ It is, in any case, not necessary to include morbidity impacts in RAINS. Environmental impacts are used by RAINS as indicators to permit optimisation against pre-defined targets. So long as the impacts selected for this process reflect changes in related effects, the use of a single impact to act as indicator is sufficient.

Table 22. Biases in the GAINS modelling. Ratings given in brackets are biases that are unlikely to have a significant effect on the outcome of the CAFE modelling.

Source of bias	Likely effect on benefit:cost ratio	Comment
Emission starting point bias for NH ₃ , NO _x , PM _{2.5} , SO ₂ and VOCs	---/+++	Negative bias arises because of uncertainty in emission inventories and the potential for switching to cleaner fuels or production systems by the baseline year for reasons unrelated to air quality regulation. Positive bias arises through uncertainty in emission inventories and possible legislative change in other areas that could cause emissions to increase.
Omission of some existing and future technical abatement measures from the RAINS model: <ul style="list-style-type: none"> • Omission of low cost measures • Omission of mid cost measures • Omission of high cost measures 	--- --- --	Biases to overestimation of costs and underestimation of the maximum feasible reduction.
Lack of account of future technical developments for existing measures:	---	Leads to an assumption of lower cost-effectiveness of existing technologies than will be achieved through further development
Omission of non-technical measures that would lead to behavioural change	---	Non-technical measures provide additional scope for emission reduction
Lack of differentiation of particles by species for health impact assessment	---/+++	Likely tendency would be to reduce the cost-effectiveness of abatement packages by inadequate focus on the most harmful particles.
Modelling urban exposure: <ul style="list-style-type: none"> • Urban background PM_{2.5} • Hot spot PM_{2.5} • Urban background ozone • Hot spot ozone 	Accounted for (---) ++ (++)	Application of the results of the CITYDELTA study enables RAINS to account for elevated background concentrations of PM _{2.5} in urban areas. However, the model does not include adjustment of data for urban background ozone, or assessment of hot-spot PM _{2.5} or hot-spot ozone. The lack of account of hot-spot conditions is considered not so important for health effect quantification as models are calibrated against background concentrations. Failure to correct for urban background ozone is of limited importance to this analysis because of the small proportion of benefits attributed to ozone and health.
Underestimation of deposition of S and N to sensitive ecosystems	--	Given that there is no economic quantification of impacts to ecosystems these impacts do not affect the reported cost-benefit relationship directly, but may influence concern over unquantified ecological impacts where stakeholders use the extended CBA to consider how unquantified effects would alter their attitude to reported cost-benefit relationships.
Overestimation of the role of N in critical loads for acidification	++	
Underestimation of ecosystem sensitivity to eutrophication	--	
Omission of health impacts on people aged under 30 years	--	When carried through to the CBA of possibly limited importance given the quantification of morbidity effects for all age groups and limited mortality amongst the under 30s in Europe.
Use of year 2000 population data and death rates for quantification of ozone effects on mortality	-	Reduces quantified impacts for future years given demographic changes that lead to an aging population in the EU27. Impact limited in CAFE because of the relatively low impact of ozone compared to PM _{2.5} .
Use of 'cut-point' for quantification of ozone impacts	-	Likely to be of limited importance.

As in any case where a large number of uncertainties are identified, there are likely to be some areas where biases cancel each other out to some degree. An obvious example from Table 22 concerns the effects of N deposition to ecosystems, where the peer review concluded that impacts of N on acidification and eutrophication were likely to be (respectively) overstated and understated.

There is a clear dominance in Table 22 of factors that bias costs up and benefits down. Overall this seems likely to lead to a bias to non-action on air pollution generally. Three questions need to be answered:

1. How large is the bias?
2. Does it apply equally to all pollutants?
3. Does it apply equally to all regions?

Quantification of the bias is clearly very difficult – if it were easy it could be incorporated into the modelling. Unfortunately there have been very few attempts to compare ex-ante estimates of control costs with actual costs. Most of the analyses that have done this have shown a strong tendency for ex-ante estimates to exaggerate costs. However, such studies are purely retrospective and cannot be used as a reliable guide to the quality of future results without further consideration. Further to this, biases will vary between pollutants, not least because of variability in the quality of emission inventories. SO₂ emissions, for example, are known with a far better level of confidence than PM emissions. Biases will also vary between regions, reflecting differences in the availability of alternative fuels, quality of national data and so on. Relevant to this discussion is the significant reduction in ammonia abatement costs introduced to the GAINS modelling in the course of EC4MACS.

Direct analysis of the effect of these biases is not possible given that they are unquantified. However, the sensitivity analysis below assesses the probability that benefits would exceed costs factoring in variation in costs in the range [GAINS estimate +20%] to [GAINS estimate - 50%]. The choice of this interval is skewed downwards (implying that GAINS is more likely to exaggerate costs than underestimate them) because of the dominance in Table 22 of factors leading to cost overestimation and the results of analysis comparing ex-ante and ex-post estimates of abatement costs (Watkiss et al, 2005 and others).

9.5.3 Biases in the benefits analysis

In common with the cost-effectiveness modelling undertaken using the GAINS model, the benefits assessment is prone to a significant number of biases. These are listed in Table 23. Readers who consider that some potential biases have been omitted from this table should consult other sections of the report to see if they are dealt with elsewhere (e.g. alternative positions on mortality valuation and aspects covered in Section 9.3 and Table 21 and Table 22 dealing with the EMEP and GAINS models respectively). Again, views on the direction and likely significance of biases are the authors' own. The omission of impacts from the analysis is dealt with below.

The failure to differentiate particles by species for the health impact assessment seems to be the most important *potential* cause of overestimation of benefits of individual measures, though it could bias results either way depending on which types of particle are targeted under a specific strategy. Other biases that could lead to overestimation seem less important, reflecting more uncertainty in data extrapolation than anything fundamental.

Table 23. Biases in the benefits analysis

Source of bias	Likely effect on benefit:cost ratio	Comment
Unquantified impacts: <ul style="list-style-type: none"> • Ecosystem acidification • Ecosystem eutrophication • Impacts of ozone on ecosystems • Damage to cultural heritage • Chronic health effects of exposure to ozone • Effects of coarse particles (size range PM_{2.5 to 10}) on health • Chronic effects of PM exposure on cardio-vascular disease • Health effects of secondary organic aerosols (SOAs) 	--- --- --- -- ---? - ---? --	Further information on the likely importance of omissions from the benefits analysis is discussed elsewhere in this report. In some cases importance will vary strongly between abatement options, e.g.; <ul style="list-style-type: none"> • An option that does not control VOCs will have very little effect on exposure to SOAs. • Abatement options controlling coarse particles could have significant additional benefits for situations where they comprise a major fraction of total particle mass.
Lack of differentiation of particles by species for health impact assessment	+++/--	Effect on quantified benefits will depend on the level of control for each type of particle.
Use of health functions from the US and western Europe	+ +/--	Further research is needed to test whether there are systematic differences between regions.
Quantification of deaths from chronic exposure to PM using techniques not based on life tables (only relevant where VSL is used for mortality valuation).	++	Some potential for double counting of deaths, depending on the time horizon used for the analysis.
Use of uniform incidence data for the whole of Europe for most morbidity effects	+ +/--	Again, further research is needed to test whether there are systematic differences between regions. The identification of consistent sets of incidence data is recognised as a problem for transferability of health response functions generally.
Use of AOT40 based relationships to quantify impacts of ozone on crops	+?	Likely to cause overestimation of impacts amongst un-irrigated crops in drier parts of Europe. Overall effect unclear.

Another area where bias is likely though it is not clear which direction that bias would go in concerns the morbidity assessment in the benefits analysis. Specifically it relates to two issues, the application of health functions from the US and western Europe across the whole of Europe, and the use of uniform incidence data. Both areas are worthy of further research. Given variability across Europe any biases that are present may tend to cancel each other out as reference to incidence data in Volume 2 of the CAFE-CBA methodology report suggests.

Despite the best efforts of the teams involved, the results of the various models are subject to a number of unquantified biases. In general terms, the most important appear to be:

- **EMEP modelling**
 - Underestimation of PM
 - Attribution of secondary organic aerosols to source (?)
- **RAINS modelling**
 - Emission starting point bias
 - Omission of some abatement techniques
 - Lack of account of future technical developments

- Lack of differentiation of particle species by effect
- **Benefits modelling**
 - Omission of impacts on ecosystems, cultural heritage, etc.
 - Lack of differentiation of particle species by effect

Several of the biases identified will have a more or less equal effect over the whole of Europe. However, the effect of others will vary from country to country. Despite the inter-linkages present between pollutants and effects, the importance of biases will also vary with the objectives defined for any scenario – a scenario focused on PM control may be little affected by sensitivities in the ozone analysis. It is thus difficult to define general rules on the reliability of the analysis for different stakeholders.

However, this does not mean that it is impossible to do anything about the biases that are present. The listing of biases in the preceding tables makes it possible for any scenario to identify which seem likely to have an important impact on the benefit:cost ratio and which are unlikely to be important. Ratings for each bias should be revised in line with the factors that influence results for any scenario. The overall impression of biases can then be considered alongside other information, for example, the probability that benefits will exceed costs for any scenario, assessed using the methods defined in Section 9.3. Given the qualitative nature of the result it is unlikely to cause a major change in policy, but it may strengthen or weaken the rationale for a specific position.

9.6. Accounting for unquantified effects

In CAFE-CBA the concept of an ‘extended CBA’ was proposed to provide further information for policy makers about the impacts that could not be quantified. It was suggested that datasheets be developed to contain the following types of information:

- Description of the impact, including components of ‘total economic value’
- Discussion of related impacts
- Confidence in attribution of impact to a specific pollutant
- Information on the distribution of impact across Europe (is it a ‘European issue’ or something to be considered at a more local level?)
- Information on importance in economic or other terms, where available (e.g. from results of willingness to pay case studies, past estimates of expenditure to deal with specific problems, etc.)

Some preliminary work was done on these datasheets, but given apparently limited interest in them the idea was put on hold. If there is sufficient interest from stakeholders it could be revived once more. Consideration was also given to the use of a formal multi-criteria assessment (MCA) framework to bring in the unquantified effects. However, it was considered unlikely that agreement would be reached on the various parameters given the diversity of stakeholders active in CAFE, so that idea was also dropped. Holland et al did, however, provide their own assessment (Table 24) of the likely significance of each unquantified effect, using a three point scale:

- ★★★ Impacts likely to be significant at the European level
- ★★ Impacts that may be significant at the European level
- ★ Impacts unlikely to be important at the European level, but of local significance
- No stars Negligible

The intention in providing information in this way was to prompt stakeholders to consider whether the impacts that have not been quantified are likely to be important enough to change the balance of costs and benefits. It is not intended that anyone should add together the star ratings given to the various impacts – they are simply intended as ‘flags’ to distinguish what is probably important from what is probably not. Stakeholders are of course free to come to their own conclusions on the relative importance of the different impacts considered in this process. Decision makers may like to consider these effects in different ways, depending on the result of the quantified cost-benefit comparison:

- In situations where costs exceed benefits:
 - Are unquantified effects likely to be sufficiently important that they would cause benefits to increase to a point where they exceed costs?
- In situations where benefits exceed costs:
 - Are unquantified effects sufficiently important that they would give much greater confidence that benefits are larger than costs?
 - Are unquantified effects large enough to have a significant impact on the ratio of benefits to costs, increasing the importance of dealing with air pollution over other problems?

Another approach that can be used is to ask how large the unquantified benefits must be for total benefits to exceed costs, or for the probability that benefits to exceed costs to be greater than some qualifying percentage. If the surplus of cost over benefit is relatively small, it may be concluded that the unquantified effects are indeed large enough to bridge the gap. If the difference is large, however, it may be concluded that they are not.

Table 24. Position on ratings for the extended CBA proposed in Holland et al (2005a). Effects considered likely to be negligible are omitted from this table.

Effect	Preliminary rating
Health	
Chronic effects of PM _{2.5} on cardio-vascular disease	★ ★ ★
Ozone: chronic effects on mortality and morbidity	★ ★
SO ₂ : chronic effects on morbidity	★
Effects of secondary organic aerosols	★ ★
Direct effects of VOCs	★
Social impacts of air pollution on health	★ ★
Altruistic effects	★ ★
Materials	
Effects on cultural assets	★ ★
Crops	
Indirect air pollution effects on livestock	★
Visible injury following ozone exposure	★
Effects of air pollution on the quality of crops, irrespective of issues concerning yield and visible injury	★ ★
Interactions between pollutants, with pests and pathogens, climate...	★ ★
Forests	
Effects of O ₃ , acidification and eutrophication	★ ★ ★
Freshwaters	
Acidification and loss of invertebrates, fish, etc.	★ ★ ★
Other ecosystems	
Effects of O ₃ , acidification and eutrophication on biodiversity	★ ★ ★
Visibility	
Change in amenity	★
Groundwater quality and supply of drinking water	
Effects of acidification	★

9.7. The TUBA Framework

This section provides the data sheets that comprise the TUBA (Treatment of Uncertainty in Benefits Analysis) Framework. The TUBA Framework seeks to bring all information on uncertainty together in a concise manner, focused on the question of whether the relationship between costs and benefits revealed by the analysis is robust. By doing so, it becomes easy to see whether the treatment of uncertainty is comprehensive or whether certain elements have been omitted. The structure used does not require full quantification. Whilst this would be preferable, it is not currently practicable, so recognising this, the framework includes review of unquantified ‘biases’. A further advantage is that the added transparency that the Framework should bring, makes it easier to challenge uncertainty analysis. This has to be a good thing given that the EC4MACS Project has already recognised the uncertainty in describing uncertainty and seems likely to be of benefit both to analysts and stakeholders.

In order to be informative, the Framework is shown completed, using the example of the final assessment undertaken as part of EC4MACS. Six cases are considered (corresponding to the six blocks on the next page), dealing with:

- CBA for all Europe, baseline vs. MFR in 2020, 2025 and 2030
- CBA for the EU27, baseline vs. MFR in 2020, 2025 and 2030

The first part of the TUBA Framework addresses the elements of uncertainty that can be addressed quantitatively. This uses information on the distribution of estimated impacts derived using the Monte Carlo analysis developed in CAFE-CBA (Holland et al, 2005b). These distributions were described having defined the variability of all inputs to the health impact assessment.

Combined with this is the sensitivity analysis that here covers different views on the valuation of mortality (identified by the ‘case names’ on the right hand side of the table). Best estimates for each sensitivity case are entered in the left hand column. TUBA then approximates the probability distribution in each case and compares against abatement costs to assess the probability of benefits exceeding costs.

The following page then reviews unquantified elements of the analysis, first identifying possible biases, then assessing which direction they are likely to drive the results. They are then weighted to identify what are considered as the most important biases. Totals (weighted and unweighted) are given at the foot of the table in order to provide an indication of the likely overall direction of bias. Given that this part of the analysis is not quantified in detail, but comes down to expert judgement, the totals shows are not definitive indicators of overall bias.

The third page of the Framework provides the conclusions of the uncertainty analysis, linking to the information on the preceding pages.

TUBA: Treatment of Uncertainty in Benefits Analysis

Quantitative assessment of uncertainty including sensitivity runs

Figures below highlighted green show where benefits > costs. Pink shading where costs > benefits.

Scenario: 2020, All countries

Abatement costs for scenario: 56

€/year Benefits	10%ile	25%ile	33%ile	50%ile	67%ile	75%ile	90%ile	Case name
125	73	90	99	115	135	149	191	Median VOLY
252	146	181	199	232	272	300	386	Mean VOLY
203	118	146	160	187	219	242	311	Median VSL
380	220	274	300	350	410	452	581	Mean VSL
96	56	69	76	88	104	114	147	Desaigues VOLY
471	273	339	372	433	509	560	721	OECD VSL

Scenario: 2025, All countries

Abatement costs for scenario: 59

€/year Benefits	10%ile	25%ile	33%ile	50%ile	67%ile	75%ile	90%ile	Case name
118	68	85	93	109	127	140	181	Median VOLY
239	139	172	189	220	258	284	366	Mean VOLY
201	117	145	159	185	217	239	308	Median VSL
377	219	271	298	347	407	449	577	Mean VSL
91	53	66	72	84	98	108	139	Desaigues VOLY
468	271	337	370	431	505	557	716	OECD VSL

Scenario: 2030, All countries

Abatement costs for scenario: 60

€/year Benefits	10%ile	25%ile	33%ile	50%ile	67%ile	75%ile	90%ile	Case name
118	68	85	93	109	127	140	181	Median VOLY
238	138	171	188	219	257	283	364	Mean VOLY
211	122	152	167	194	228	251	323	Median VSL
395	229	284	312	363	427	470	604	Mean VSL
92	53	66	73	85	99	109	141	Desaigues VOLY
490	284	353	387	451	529	583	750	OECD VSL

Scenario: 2020, EU27

Abatement costs for scenario: 38

€/year Benefits	10%ile	25%ile	33%ile	50%ile	67%ile	75%ile	90%ile	Case name
62	36	45	49	57	67	74	95	Median VOLY
123	71	89	97	113	133	146	188	Mean VOLY
105	61	76	83	97	113	125	161	Median VSL
195	113	140	154	179	211	232	298	Mean VSL
49	28	35	39	45	53	58	75	Desaigues VOLY
241	140	174	190	222	260	287	369	OECD VSL

Scenario: 2025, EU27

Abatement costs for scenario: 40

€/year Benefits	10%ile	25%ile	33%ile	50%ile	67%ile	75%ile	90%ile	Case name
59	34	42	47	54	64	70	90	Median VOLY
115	67	83	91	106	124	137	176	Mean VOLY
104	60	75	82	96	112	124	159	Median VSL
193	112	139	152	178	208	230	295	Mean VSL
46	27	33	36	42	50	55	70	Desaigues VOLY
238	138	171	188	219	257	283	364	OECD VSL

Scenario: 2030, EU27

Abatement costs for scenario: 41

€/year Benefits	10%ile	25%ile	33%ile	50%ile	67%ile	75%ile	90%ile	Case name
59	34	42	47	54	64	70	90	Median VOLY
115	67	83	91	106	124	137	176	Mean VOLY
109	63	78	86	100	118	130	167	Median VSL
202	117	145	160	186	218	240	309	Mean VSL
46	27	33	36	42	50	55	70	Desaigues VOLY
251	146	181	198	231	271	299	384	OECD VSL

Bias assessment for unquantified uncertainties				
The views expressed here are those of the author.				
	Direction of bias		Weight	Comments
	Costs up or impacts down?	Costs down or impacts up?		
Emissions				
Energy	0	0	0%	Associated emissions from the energy sector are well researched with no evidence of any significant bias.
Transport	0	0	0%	As above
Agriculture	0	0	0%	As above
Dispersion modelling				
Ozone concentrations	0	0	0%	Overall, assumed that average concentrations are reasonable, with no systematic bias.
PM2.5 concentrations	0	0	0%	As above
Impacts				
Health	1	0	50%	Believed that some health impacts are excluded, moderated by concern over some valuations possibly being too high, e.g. for chronic bronchitis.
Materials	0	0	0%	Utilitarian material damage makes minimal contribution to overall effects
Materials in cultural heritage	1	0	20%	Low urban SO2 levels suggest this is of limited importance
Crops	0	0	0%	Accounted for in the modelling
Other agriculture	0	0	0%	Considered unimportant
Ecosystems	1	0	100%	Not included. Extent of exceedance of eutrophication in particular suggests that this effect is significant.
Cobenefits and trade offs				
Greenhouse gas emissions	1	0	10%	Small effect on GHG emissions unaccounted for
Discharge to water (not for treatment)	0	1	10%	Considered unimportant as plant will operate within discharge consents. However, linked with solid waste arisings (see below)
Discharge to water treatment system	0	0	0%	Again, discharge consents apply
Other pollutant emissions	1	0	1%	Heavy metal and other unaccounted for emissions considered trivial.
Solid waste generation	0	1	100%	Long term capacity for dealing with solid waste generated by abatement options is unclear.
Noise	0	0	0%	No change in noise identified.
Other cobenefits, trade offs	0	0	0%	No other cobenefits and trade offs identified.
Cost data				
Costs to agriculture	?	?	?	Direction of bias unclear
Costs to transport	1	0	100%	Potential for overestimation of costs as a result of improved efficiency of technologies over time, and effects of climate policy
Costs to industry	1	0	100%	As transport
Costs to domestic users	?	?	?	Direction of bias unclear
Costs to commercial users	1	0	50%	As transport
Un-weighted total	8	2		
Weighted total	4.31	1.1		

Conclusions

[Opening comment: A marginal CBA carried out directly to inform the EU's policy making process would focus attention down to a smaller region of the cost curve for identification of the most efficient and justifiable levels of abatement. Here, however, this is not possible as analysis has considered only baseline and MFR scenarios, so results are taken as they stand, to demonstrate use of the TUBA framework. It must be recognised, however, that there will be very high positive benefit-cost ratios for some measures within the MFR 'package' and very low ratios for others.]

The analysis of uncertainties has quantified the possible consequences of variability with respect to data used for (e.g.) response functions and valuation, and sensitivity to key assumptions. It has also sought to provide a comprehensive overview of unquantified biases that affect the results, focusing on the likelihood that the benefits of action will exceed costs. By accounting for these various elements the analysis of uncertainties provides a comprehensive overview of the robustness of results.

Results of the quantitative uncertainty analysis indicate that in all cases where analysis considers all European countries, there is at most a 10% probability that benefits would not exceed costs. In all cases using a VOLY higher than from Desaigues et al (2011) or using the VSL, the probability of costs exceeding benefits is less than 10%. Restricting analysis to the EU27 reveals a slightly different pattern. There is around a 10% probability of costs exceeding benefits when mortality is valued with the median VOLY and a 25 to 40% probability when the Desaigues et al VOLY is applied. Again, there is substantially less than a 10% probability of a net cost when applying the VSL. Overall, results consistently indicate that the move to the MFR scenario would be beneficial to society on economic grounds. *[Leaving aside the non-marginal nature of the scenario comparison, for the purposes of illustration.]*

The bias analysis indicates that (in the opinion of the author) there are more biases that either reduce estimates of benefits or increase estimates of cost than vice versa; in other words the biases overall appear likely to act against any package of measures passing a cost-benefit test. The weighted equivalent (again, here based on the author's views, rather than, e.g. an expert panel though there is no reason that such a panel could not be convened in the future) provides a very similar outcome. Accepting the views expressed in the bias analysis as a reasonable representation of reality would suggest that the conclusion above that benefits are likely to exceed costs becomes more robust when additional uncertainties are brought into the equation.

A caveat in relation to the bias analysis concerns those elements where it was concluded that the direction of bias was unclear (these concerned costs to agriculture and costs to domestic users). Here and elsewhere, the robustness of the analysis could be improved through further discussion and data collection if it was thought necessary.

[Closing comment: The material presented here indicates how uncertainty analysis to support cost-benefit analysis can be structured in such a way as to provide an overview of the uncertainties affecting the results. It is kept intentionally brief in order to assist stakeholders develop such an overview. Alternative views and feedback are welcome.]

9.8. Discussion

If the costs and benefits of air pollution control were known with absolute confidence there would be no problem in comparing the two. However, costs and benefits are subject to uncertainties and some of them (on both sides of the cost-benefit equation) are significant. The quality of knowledge for identification of these uncertainties is variable, as is the availability of quantitative data with which to describe them. Further to this, some uncertainties are statistical and continuous in nature, some relate to discrete choices (e.g. selection of approaches for the valuation of air pollution – related mortality) whilst some simply relate to a lack of knowledge. It is clear from this that the development of a fully consistent approach to description of uncertainty across the CAFE analysis is not straightforward.

The extent to which uncertainty needs to be considered in any situation is largely dependent on the balance of costs and benefits. Where estimated costs far exceed estimated benefits it is unlikely that any assessment of uncertainty would change the perception of that relationship unless some possible outcomes were politically untenable (an obvious example, though not one directly relevant to EC4MACS, concerns major nuclear accidents). Similarly, where benefits far exceed costs, uncertainties should be of limited importance.

Consideration of uncertainty in comparison of costs and benefits cannot, therefore, be an automatic process. Awareness needs to be raised of the component uncertainties of each part of the analysis. This has been addressed in this report. The most important of the component uncertainties should be highlighted and quantified to the extent possible. Again, this is done here. Consideration also needs to be given to how satisfactory the assessment of uncertainty is. We believe that this report lays the ground for a good quality assessment of uncertainty, though this question needs to be asked against analysis of specific scenarios.

This report has identified three main strands for assessment of uncertainty, these being statistical analysis, sensitivity analysis and assessment of biases, the latter being largely associated with gaps in knowledge. Some of these can be addressed relatively easily in quantitative terms. Others cannot, and require a more subjective assessment. Irrespective of whether they can be addressed quantitatively or semi-quantitatively, all of the uncertainties identified here are potentially important and need to be considered. This Chapter has also raised the problem that given the multi-component nature of the EC4MACS modelling framework, consideration of uncertainties in the CBA can be extremely long-winded (being the final part of the analysis, it is affected by uncertainties in all of the other models). This has led to the development of the TUBA Framework, designed to report uncertainty in a reasonably concise way, and to focus its reporting on potential consequences for the robustness of conclusions drawn.

Whilst it is clear that there are a large number of uncertainties that affect the analysis of scenarios being considered in the EC4MACS Project it is the view of the authors that this is not a barrier to effective and efficient decision making, because:

- We know a lot about the uncertainties that are present.
- We have a range of tools for assessment of these uncertainties.
- We can use these tools to see how uncertainty could influence the reported relationship between costs and benefits.
- We now have a concise system for summarising information on uncertainty.

It is worth considering the objective of cost-benefit analysis, namely to identify approaches that represent least cost to society. ‘Cost’ here includes environmental and health costs as well as pollution abatement costs. Rabl et al (2005) focused on the effect of uncertainty in determining the least cost position. They concluded that for continuous choices such as the development of emission ceilings for sectors or regions, the cost penalty turns out to be “*remarkably insensitive to error*”. They observed that an error of a factor 3 up or down in damage estimates for NO_x and SO₂ would potentially increase the social cost by at most 20% and in many cases much less. The costs analysis used for the paper was based on the earlier RAINS cost curves, and so is particularly relevant to EC4MACS. Much of the reason for this insensitivity rests in the non-linear shape of the marginal abatement cost curve.

10. Summary

This report provides an overview of the methodology used for quantification of the benefits of measures considered under EC4MACS, and for comparison of costs and benefits.

Quantification of impacts and monetisation should ideally follow the impact pathway approach, as it traces a logical and sequential path from cause to effect. Application of this approach is limited by the availability of data, for example on stock-at-risk, exposure-response and valuation. Hence, whilst it is possible to quantify some damages to health, crops and materials, it is not possible to quantify all impacts. Fortunately, there are grounds for believing that the impacts that can be quantified are amongst the most significant. However, some impacts that are omitted from quantification are important, the most obvious omission being effects on ecosystems from deposition of acidifying pollutants and nitrogen.

Consideration needs to be given to how these omitted effects can be brought to quantification – indeed, some ideas are presented in relation to ecosystems, drawing and will be further developed through the ECLAIRE Project. In the continued absence of quantification, however, attention should be given to approaches for ensuring that unquantified effects have sufficient prominence that they are not forgotten when decisions are made. The TUBA Framework provides a possible mechanism for ensuring this.

Going beyond EC4MACS, it is intended to keep the methods presented here under review and provide updated methodology reports when possible. In the short term (mid to late 2013) there will be updated recommendations for health impact assessment. In the medium term, further analysis of ecosystems should become possible.

Finally, it should be stressed that the methods presented here are not solely for use in conjunction with GAINS and the overall EC4MACS modelling framework. The original CAFE-CBA approach has been used for other purposes, including consideration of IPPC (see EIPPC Bureau, 2006) and assessment of the burdens posed by large point sources (e.g. EEA, 2011).

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Appendix 1: Abbreviations and Terminology

AF _{st} 6	Accumulated stomatal flux of ozone above a threshold of 6 mmol.m ⁻²
ALPHA2	Atmospheric Long-range Pollution Health/Environment Assessment Model version 2
AOT40	Accumulated concentration of ozone over a threshold of 40 ppb
ASTM	The degradation of coatings measured according to ASTM D 1150-55, 1987, giving a range between 1 and 10 where 10 corresponds to an unexposed sample.
CAFE	Clean Air For Europe
CAPRI	Common Agricultural Policy Regional Impact Analysis Model
CBA	Cost-benefit analysis
CBI	Confederation of British Industry
CCE	Coordinating Centre for Effects
CHA	Cardiac Hospital Admission
CLRTAP	Convention on Long Range Transboundary Air Pollution
CO	Carbon Monoxide
CO ₂	Carbon dioxide
COI	Cost of illness
C-R	Concentration-response (function)
CV or CVM	Contingent valuation (method)
CVA	Cerebro-vascular conditions
DEFRA	UK Department for Environment, Food and Rural Affairs
EC	European Commission
EC DG ENV	European Commission Directorate General Environment
EIPPC	European Bureau on Integrated Pollution Prevention and Control
EMEP	The Cooperative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe
ERV	Emergency Room Visit
EU	European Union
FAO	(UN) Food and Agriculture Organization
FGD	Flue gas desulphurisation
GAINS	Greenhouse Gas and Air Pollution Interactions and Synergies Model
GBD	WHO Global Burden of Disease Project
GDP	Gross Domestic Product
GHG	Greenhouse gas
H ⁺	Hydrogen ion
HF	Hydrogen fluoride
HIA	Health Impact Assessment
IIASA	International Institute for Applied Systems Analysis
ICP	International Cooperative Programme
ICP/MM	International Cooperative Programme on Mapping and Modelling
IOM	Institute of Occupational Medicine
IPPC	Integrated Pollution Prevention and Control
LRTAP	Convention on Long Range Transboundary Air Pollution
MCA	Multi-criteria assessment
MTFR	Maximum feasible reduction scenario
MRAD	Minor restricted activity day
NEBEI	Network of experts on benefit and economic instruments

NECD	National Emission Ceilings Directive
NEEDS	New Energy Externalities Development for Sustainability
NEWEXT	New Elements for the Assessment of External Costs from Energy Technologies
NH ₃	Ammonia
NH ₄ ⁺	Ammonium ion
NO	Nitrogen monoxide
NO ₂	Nitrogen dioxide
NO ₃ ⁻	Nitrate
NOAA	U.S. National Oceanic and Atmospheric Administration
NO _x	Oxides of nitrogen
O ₃	Ozone
PEC	Particulate elemental carbon
PM ₁₀	Fine particles less than 10 µm in diameter
PM _{2.5}	Fine particles less than 2.5 µm in diameter
PRIMES	European energy systems model
PRTP	Pure Rate of Time Preference
PVC	Poly-vinyl chloride
RAD	Restricted Activity Day
RAINS	Regional Air Pollution Information and Simulation
RHA	Respiratory Hospital Admission
RY	Relative yield
SEI-York	Stockholm Environment Institute at York
SO ₂	Sulphur dioxide
SO ₄ ²⁻	Sulphate
TFEAAS	Task Force on Economic Aspects of Abatement Strategies
TFH	Task Force on Health
TREMOVE	European transport, energy and environment model
UNECE	United Nations Economic Commission for Europe
URS	Upper Respiratory Symptom
USEPA	United States Environmental Protection Agency
VOCs	Volatile organic compounds
VOLY	Value of life year
VSL	Value of statistical life
WHO	World Health Organization
WLD	Work Loss Day
WTA	Willingness to accept
WTP	Willingness to pay
YOLL	Years of life lost

MATHEMATICAL NOTATION

The following prefixes and suffixes are used in this work;

Ex, E- x as a suffix to a number, denotes that the number in question should be multiplied by 10 to the power x or $-x$. Hence 6.4E-3 is equal to 0.0064.

The following prefixes to units are also used;

n = nano = 10^{-9}

μ or u = micro = 10^{-6}

m = milli = 10^{-3}

k = kilo = 10^3 = thousands

M = mega = 10^6 = millions

G = giga = 10^9 = billions

This system is standard notation in the sciences. Note that m and M are not equivalent (by a factor of 10^9) and hence should not be interchanged.

Appendix 2: Worked examples of impact assessment

Two worked examples are provided, both concerning health impact assessment. The first deals with effects of PM_{2.5} on the development of chronic bronchitis and the second, effects of ozone on respiratory hospital admissions. Both cases quantify for a hypothetical grid cell containing 100,000 people subject to an exposure of 10 µg.m⁻³.

Effect: Development of chronic bronchitis in adults aged >27

Original function: 7.0% change in baseline attack rates, per 10 µg/m³ PM₁₀

	Quantity	Units	
Step 1: Quantify total population exposure in grid cell x, y			
A	Population in grid cell x, y	100,000	people
B	PM _{2.5} concentration in grid cell x, y	10	µg.m ⁻³
C	Exposure in grid cell x, y (A × B)	1,000,000	person.µg.m ⁻³
Step 2: Adjust exposure for adult population aged >27			
D	Fraction of total population aged over 27	0.66	no units
E	Adjusted exposure (C × D)	660,000	person (aged>27).µg.m ⁻³
Step 3: Adjust concentration-response function to account for differences in pollution metrics			
F	Original function	7%	change in baseline attack rates, per 10 µg.m ⁻³ PM ₁₀
G	Factor to convert from PM ₁₀ to PM _{2.5}	1.54	no units
H	Factor to convert from 10µg.m ⁻³ to 1µg.m ⁻³	0.1	no units
J	Revised function (F × G × H)	1.08%	change in baseline attack rates, per 1 µg.m ⁻³ PM _{2.5}
Step 4: Quantify relevant incidence rates			
K	Incidence rate of chronic bronchitis	0.707%	%
L	Non-remission rate	53.40%	%
M	Adjusted incidence rate	0.378%	%
Step 5: Quantify impacts			
N	Increase in incidence in grid cell x, y (E × J × M)	26.9	New cases of chronic bronchitis in grid cell x, y
Step 6: Value impacts			
O	Value per case of chronic bronchitis	€ 190,000	€
P	Economic value in grid cell x, y (N × O)	€ 5,100,000	€
Step 7: Aggregate across other grid cells			
Calculate national totals			
Calculate regional (EU15, EU27, etc.) totals			

Effect: Respiratory hospital admissions (RHAs) in the general populationOriginal function: 0.3% change in admissions per $10 \mu\text{g}/\text{m}^3 \text{O}_3$ (8-hr daily average)

	Quantity	Units
Step 1: Quantify total population exposure in grid cell x, y		
A Population in grid cell x, y	100,000	people
B O_3 concentration in grid cell x, y	10	$\mu\text{g}.\text{m}^{-3}$
C Exposure in grid cell x, y ($A \times B$)	1,000,000	$\text{person}.\mu\text{g}.\text{m}^{-3}$
Step 2: Adjust concentration-response function to account for differences in pollution metrics		
D Original function	0.3%	change in baseline admissions per $10 \mu\text{g}.\text{m}^{-3} \text{O}_3$
E Factor to convert from $10 \mu\text{g}.\text{m}^{-3}$ to $1 \mu\text{g}.\text{m}^{-3}$	0.1	no units
F Revised function ($D \times E$)	0.03%	change in baseline admissions, per $1 \mu\text{g}.\text{m}^{-3} \text{O}_3$
Step 3: Quantify relevant incidence rates		
G Incidence rate for RHAs in the general population	0.617%	%
Step 4: Quantify impacts		
H Increase in incidence in grid cell x, y ($C \times F \times G$)	1.9	Additional respiratory hospital admissions in grid cell x, y
Step 5: Value impacts		
J Value per respiratory hospital admission	€ 2,141	€
K Economic value in grid cell x, y ($H \times J$)	€ 3,700	€
Step 6: Aggregate across other grid cells		
Calculate national totals		
Calculate regional (EU15, EU27, etc.) totals		