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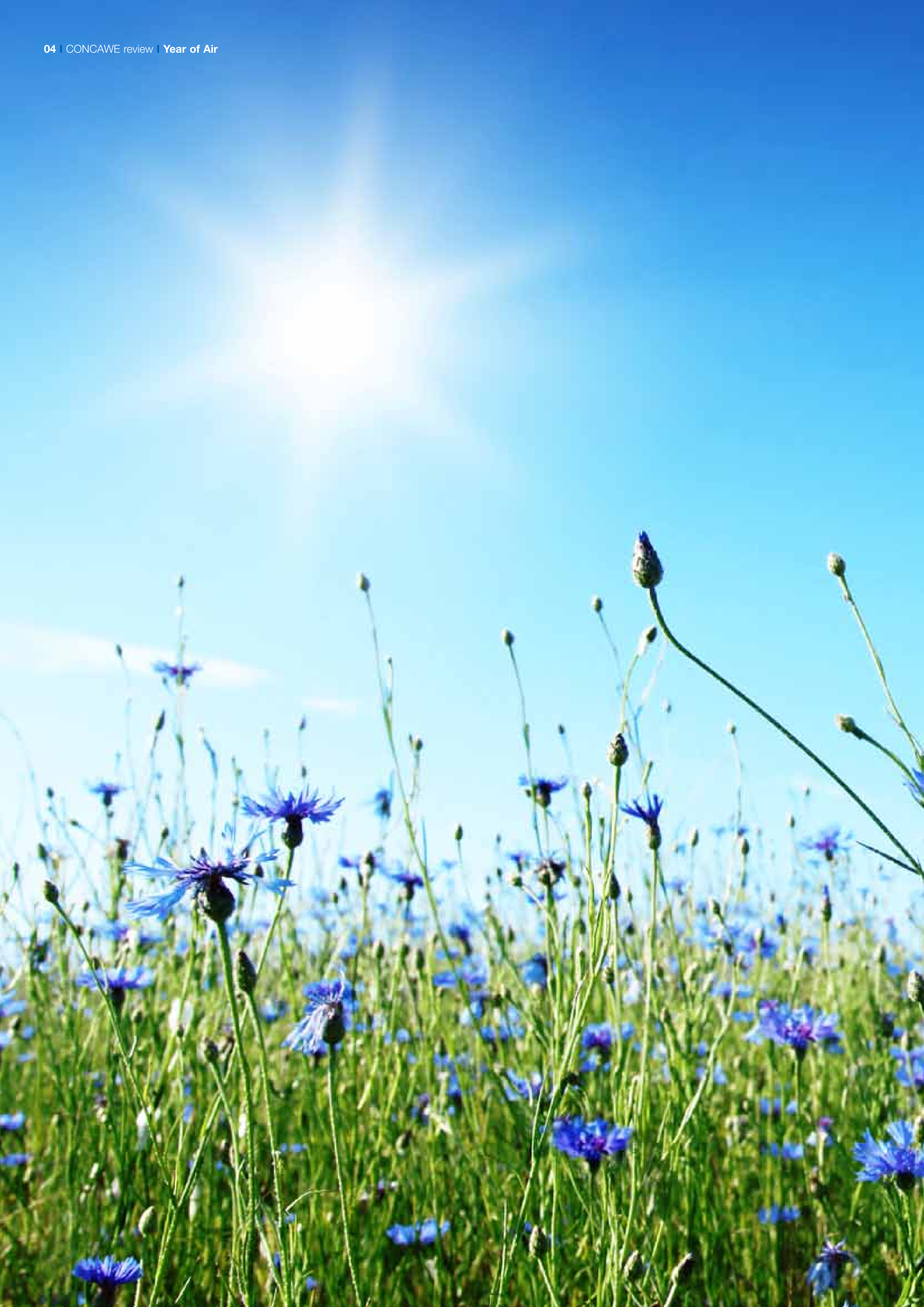


1 - Preface

The Thematic Strategy on Air Pollution (2005) has been under review since 2011 with the first of five stakeholder meetings taking place in January 2012. The underpinning work is very complex and relies heavily on modelling studies. CONCAWE as an active stakeholder has participated in all of the formal engagement meetings and performed its own analyses working from two viewpoints. Firstly to understand the work taking place, and secondly to contribute insights that should lead to more robust policy conclusions being drawn. This work has been reported via the CONCAWE Review (CONCAWE 2012a, CONCAWE 2012b) and technical contributions to the Stakeholder consultation[1].

The aim of this special issue of the CONCAWE Review is to bring together in a single document all recent information that CONCAWE has generated in the field of the integrated assessment modelling and cost-benefit analysis related to air policies[2], with a special focus on the uncertainties/sensitivities and their implications for the policy-making process. Some of the information is published for the first time in this issue (e.g. on ecosystem services).

To make this material more accessible to non-specialists in the field, at the beginning of each subsection there is a short summary of the main outcomes of each CONCAWE study and where possible and appropriate this has been expressed in simplified form.

[1] The original contributions to the Stakeholder consultation are included in the electronic version of this Special Review which can be found on CONCAWE website www.concawe.eu.

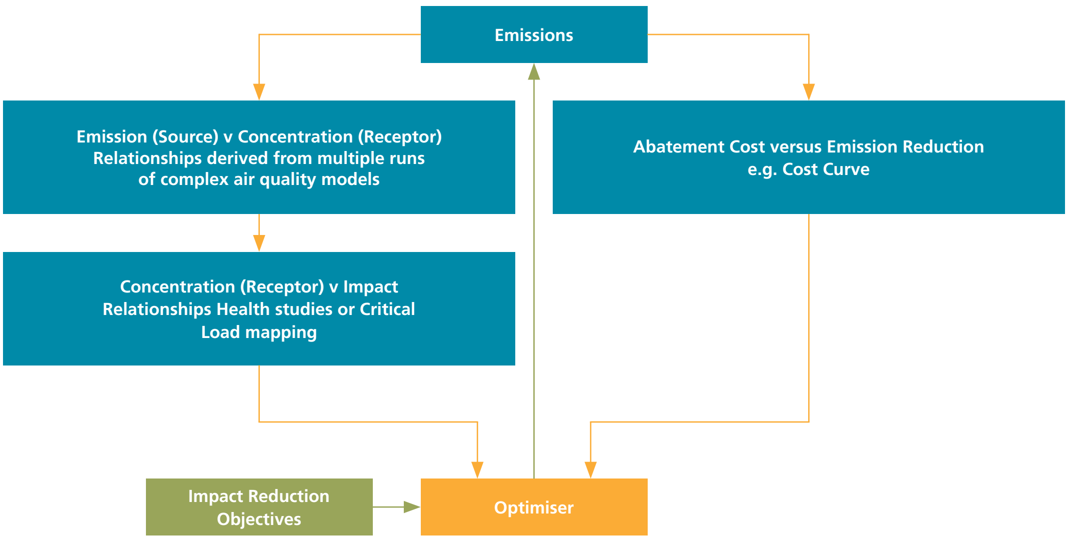
[2] The technical work described in this document was completed in the autumn of 2013 before the adoption of the European Commission Clean Air Policy Package.

2 - Executive summary

Developing policy, which entails legally binding commitments, on the basis of a model prediction of future emissions, requires very careful consideration of the sensitivities of the model to the assumptions made about the future world. The development of economic and industrial activity in Europe over the next 10-15 years is subject to significant uncertainties. The use of a single energy scenario with a limited sensitivity analysis to develop legislative proposals is therefore unwise. To draw attention to this CONCAWE has explored several important sensitivities demonstrating how, under different plausible future scenarios, the implementation costs of high ambition levels may increase significantly or emission reductions become unfeasible.

Integrated Assessment Modelling (IAM) is an important tool used to inform policy makers on the scope for further emission reduction options, taking into account baseline emissions (current legislation), associated impacts and cost-effective emission reduction strategies.

Figure 1
Simplified Integrated Assessment Modelling Framework



The aim of this special issue of CONCAWE Review is to bring together in a single document the information that CONCAWE has generated in the field of the integrated assessment modelling and cost-benefit analysis during the period of stakeholder consultation (June 2011- September 2013). This issue has a particular focus on the uncertainties/sensitivities involved and their implications for the policy-making process. While most of the material is drawn from previously published work (CONCAWE 2012a, CONCAWE 2012b and contributions to the Air Policy review stakeholder consultation[3]), some of the information is published for the first time (e.g. ecosystem services).

The illustrative sensitivity analysis described in **Section 4 "Uncertainties under the microscope"** addresses six key issues using CONCAWE's in house Integrated Assessment Model:

Policy vulnerability to under-delivery of Euro VI/6 NOx emission reductions:

Policy scenarios leading to revised Thematic Strategy on Air Pollution (TSAP) targets and national emission ceilings must account for uncertainties in the reductions in road transport NOx emissions associated with the introduction of Euro VI/6 standards in 2014/17. In the past, real world NOx emissions from the road transport sector have been substantially greater than forecast from the regulated emission limits (from Euro II/2 to Euro V/5). This has led to substantial problems in achieving obligations under the current National Emission Ceiling Directive (NECD) and Ambient Air Quality Directive (AAQD) in a number of Member States.

Sensitivity analysis (based on the energy scenario generated by the PRIMES 2009 model) shows that if, under real life driving conditions, EURO VI only delivers a 50% improvement over Euro V; and Euro 6 delivers no improvement over Euro 5 (versus the eightfold and twofold NOx/km reductions respectively assumed in the Air Policy review process) then, at a high ambition scenario, the cost for non-road transport sectors could rise from 7 to 20 b€/year to achieve the PM impact reduction target.

[3] The original contributions to the Stakeholder consultation are included in the electronic version of this Special Review which can be found on CONCAWE website www.concawe.eu.

Policy dependency on NH_3 emission reductions from Agriculture:
Reduction of emissions of ammonia is central to cost-effective reductions in human health exposure to particulates. The sensitivity analysis (based on PRIMES 2009) shows that the cost of a high ambition level of reduction of PM impact on human health increases by a factor of nearly 5 if NH_3 emissions are not reduced beyond the baseline. Neither is it possible to meet ambitious acidification or eutrophication targets if ammonia emissions are not reduced.

Policy needs to consider multiple time horizons,
The reduction of emissions from already agreed legislation, together with structural changes (e.g. changing energy use in the baseline energy scenario), has significant effects on emissions with time. Investing heavily in abatement technology to achieve emissions reductions that will be reached by other means just a few years later could lead to unnecessary additional financial pressures and regret investment.

Policy vulnerability to a single energy scenario:
Accounting for the uncertainties in defining the 'future world' is vital to ensure that ambition levels (expressed as revised national emission ceiling commitments) do not result in significant escalation in compliance costs or non-achievability in a different actual future energy world. CONCAWE sensitivity analysis comparing the annual abatement costs for stationary sources for PRIMES 2009 and for the National Energy Scenarios indicates that the costs to comply with high ambition levels in PRIMES 2009 increase by three times when National Energy Scenarios are considered. When implied measures are close to or at Maximum Technically Feasible Reduction (MTFR), individual pollutant ceilings based solely on a single PRIMES scenario would be unattainable under an alternative energy scenario. The current difficulties in some Member States in meeting 2010 NOx ceilings illustrate the vital need to include such energy uncertainties in policy development.

Policy benefit of more fully accounting for Short Lived Climate Forcers (SLCF):
Attributing a CO_2 credit or debit to SLCF emissions (based on carbon price) and including them in the optimization strategy can give an entirely different perspective to control policies. Due to the cooling effect of SO_2 emissions and the warming effect of black carbon emissions the inclusion of this factor in the optimisation shifts the policy emphasis away from NOx and SO_2 controls on stationary sources to focus on PM emissions, even at relatively low carbon prices and long-time horizons.

Policy implications of differentiating the toxicity of primary and secondary components of the overall PM mix:
Despite the recent review of evidence led by the World Health Organisation (WHO) in the project REVIHAAP, (WHO, 2013), the WHO has not yet provided guidance on how to differentiate the impacts of the different components of the PM mix e.g. primary and secondary components. As a consequence, currently all PM components are given 'equal impacts potency'. CONCAWE sensitivity analysis shows that when both differentiated toxicity and SLCF are accounted for in the optimization strategy, even at a modest differentiated toxicity assumption, there is a profound change to the resulting package of measures. The outcome is a control strategy focused on primary particle emissions that results in a significantly lower additional cost to achieve a given PM health impact reduction target.

Section 5 "CBA under the microscope" explores several inputs to and key assumptions of the cost-benefit analysis (CBA) methodology developed under the Clean Air For Europe (CAFE) programme and used for the Air Policy review. This section shows that:

Loss of statistical life expectancy is the only appropriate metric to use in assessing the chronic effect of PM exposure on human health. The use of premature deaths as a metric for PM chronic exposure is not appropriate because attribution of cause of death to air pollution is not possible.

The Value Of a Life Year (VOLY) should be derived from Willingness To Pay (WTP) studies that directly elicit the value of changes in life expectancy. A new methodology to derive a single VOLY is also presented in this section, which respects the individual expressions of willingness to pay in a given survey. The VOLY value derived using this new methodology is significantly lower than the one used from CAFE in the current Air Policy review process. This more robust VOLY value leads to a reduction of benefits of chronic mortality by a factor of 6 compared to the values derived from the NewExt[4] study (adjusted for inflation), that are considered in the CBAs supporting the Air Policy review process.

[4] NewExt is one of a series of studies to assess the external costs of the energy sector. Its findings were used in the Cost Benefit Analysis supporting the 2005 CAFE program (NewExt 2004)

The recent CBAs informing the current policy review process for monetising morbidity impacts result in overvaluation by at least a factor of two.

The overall valuation of health effects mainly from chronic exposure to PM is overvalued by around a factor of 4 in the CBAs supporting the Air Policy review process.

It is also regrettable that the CBA was not appropriately extended to explore whether there are other "risks" to which, if the same expenditure was spent, would return a greater societal benefit.

The consequence of these changes in valuations is that the point at which marginal costs exceed marginal benefits would lie at a significantly lower PM gap closure[5] than the 75% used as central ambition level for health impacts from PM in IIASA report #10. (IIASA 2013).

It is important to note that IIASA Report #10 follows a very different approach compared to the CAFE study. In the CAFE study, costs and benefits of different scenarios were compared, but in IIASA Report #10 marginal costs and benefits are compared in order to find the optimal gap closure. With this change in approach the concern is that the uncertainties in the benefit figures are very high, and the costs curves are very steep for high policy ambition levels. This means that small uncertainties in costs or benefits can significantly change the point of optimal gap closure, as demonstrated in this section.

Section 6 "Ecosystem services under the microscope" examines the current state of the art in this relatively new field recognising that to date only some estimated crop yield losses have been included in monetisation of such benefits in the European policy arena. Section 6 shows that:

The effect of complex ecosystem dynamics needs to be better understood prior to including ecosystem impacts in CBA models. In particular, there are likely to be complexities resulting from interactions between different stressors including air pollution on ecosystems. In addition, there may be lag effects occurring, reducing air pollution deposition rates to below critical load levels may not immediately lead to restoration of ecosystem functioning or ecosystem services supply. These lag effects affect the cost benefit ratio of different policy options.

There is a need to examine how marginal costs and benefits of changes in ecosystem services supply resulting from changes in air pollution can be analysed. An important question is what effect passing critical load thresholds will have on ecosystem functioning and subsequently the supply of ecosystem services. This effect is likely to differ for different ecosystem types and different types of ecosystem services.

There is a need to better understand society's willingness to pay for biodiversity. Reducing eutrophication, in particular, may lead to lower timber production and lower carbon sequestration in nitrogen limited forest ecosystems, but may enhance biodiversity in these forests. A question is how biodiversity effects and negative impacts on other services can be compared.

[5] Gap Closure: the reduction in impacts, expressed as a percentage, of the maximum further impact reduction achievable in moving from Current Legislation scenario to Maximum Technical Feasible Reduction

3 - Science and Policy interface: role of integrated assessment modelling (IAM) and cost-benefit analysis (CBA)

Integrated Assessment Modelling (IAM) has been at the heart of European air quality policy development for more than two decades[6]. Such tools provide a useful framework for policy makers to connect the increasingly complex science dealing with multiple pollutants and multiple effects to practical and cost-effective policy.

Given that the complexity of the underlying science is embedded (often deeply embedded) within the IAM, the development and use of such tools places significant responsibility on the scientific community involved. First they need to ensure that 'good science' is incorporated into the model and that uncertainties in the science are made transparent and their policy relevance explored. Second, they need to ensure that the exogenous or endogenous data driving the model accounts for uncertainties (e.g. alternative 'future worlds'). Complex science connected to practical policy 'at the push of a button' is alluring since it no longer requires stakeholders (especially hard pressed policy makers) to invest in understanding the science or its limitations. The danger is that all this complexity becomes a black box where only the inputs and outputs are visible.

This said, in the ever complex 'multi-pollutant', 'multi-issue' world of air quality, IAMs are vital to the development of practical policy but must be appropriately deployed. In principle such tools enable the full 'policy envelope' to be explored and provide an ideal framework to explore the influence of 'uncertainties' and express them in policy terms.

In this Review we have brought together the results of a comprehensive range of such uncertainty or sensitivity scenarios to illustrate how vital it is to fully utilize the capabilities of IAM tools to ensure policies are robust.

We extend our Review to the topic of Cost Benefit Analysis (CBA). Whether appropriate or not, CBA has increasingly been used by policy makers to 'justify' the proposed ambition level of air quality policies in Europe. Therefore, as for IAM, CBA needs to be based on sound science which accounts for alternative views and for the uncertainties both in the valuation of external costs and in the impacts that are being valued. This is well illustrated in the range of valuations that result from Willingness To Pay (WPT) surveys used as the main 'valuation input' for determining the external cost for long term PM impacts on human health. Survey data shows some three orders of magnitude variations in individual responses (discounting the zero/close to zero responses). The distribution is also highly skewed to the low end valuations.

The use of survey data to value the external costs also brings with it an inherent difficulty since, unlike market surveys which seek to provide data on willingness to pay for a new product launched onto the market, the actual willingness to pay is never tested. This, in itself, suggests reliance on a single WTP for policy development is far from robust.

In addition, it is important to note that policy is rightly shaped by many factors. CBA is by its nature 'single issue' focused. What it does not tell the policy maker is whether the expenditure on this 'societal risk' if spent on another 'risk' would return a greater societal benefit. This need to spend proportionally across a range of risks is often lost in CBA. This is particularly relevant for the case of air pollution control since health benefits can be generated on the basis of a range of different policies and a CBA of air pollution control policies does not identify which of the options promoting better health is the most cost-effective.

In this Review we explore these concerns and their implications in the development of a robust policy. We also set forth what we believe might be a better approach. In addition, recognizing the growing body of work around valuing ecosystem services, we provide what we hope are some first thoughts that are relevant in considering ecosystem impacts in air policy assessments.

[6] The significant shift to an effects based approach to European air quality policy took place in the early part of the 90's and was first deployed (including IAM) in the technical work underpinning the UN-ECE second sulphur (Oslo) Protocol. The availability of robust European Scale air quality models such as EMEP together with the comprehensive mapping of critical loads for ecosystems paved the way for IAM. Here simplified emission-impact relationships based on the results of European scale air quality modelling are integrated in a framework with emission control measures and their costs to enable optimum, cost-effective policies to be explored. The IAM studies have been centered on the RAINS/GAINS model developed and maintained by IIASA.

Gothenburg Protocol (GP):

The Convention on Long-range Trans-boundary Air Pollution (CLTAP) was adopted in November 1979 within the framework of the Economic Commission for Europe on the Protection of the Environment. There are currently 32 Signatory countries to the Convention including most western European countries, Canada, the Russian Federation, Ukraine and the USA. A total of 51 countries are party to the Convention. The Convention includes eight protocols that identify specific obligations to be taken up by the signatory parties. The Gothenburg Protocol was signed in 1999 in Gothenburg and entered into force in 2005. It sets emissions ceilings for sulphur dioxide, nitrogen oxides, volatile organic compounds and ammonia in order to reduce acidification, eutrophication and ground-level ozone. In the EU, the Gothenburg Protocol has been implemented through the National Emission Ceilings (NEC) directive. In the Gothenburg Protocol, emission limits are set for each participating country.

Substantial amendments to the Gothenburg Protocol were agreed in May 2012. These amendments included new commitments for the reduction of $PM_{2.5}$, specific attention for black carbon as driver for both air pollution and climate change, and new commitments to reduce the emissions of sulphur dioxide, nitrogen oxides, ammonia, and volatile organic compounds. In addition, a number of new countries signed up for the Gothenburg Protocol, or indicated their interest in becoming signatories, including Russia and Belarus. Substantial improvements in air quality can be expected as result of the implementation of the revised Gothenburg Protocol.

Thematic Strategy on Air Pollution (TSAP)

The Thematic Strategy on Air Pollution (TSAP) (September 2005) is one of the seven thematic strategies in the Sixth Environmental Action Programme adopted by the EU in 2002. It supplements national and preceding EU legislation by establishing objectives for air pollution and proposing air pollution control measures. The TSAP covers a wide range of air quality issues and potential pollutants, with a focus on Particulate Matter.

Basic concepts in IAM language

CLE: Current legislation. Usually used to refer to the emissions that result from current legislation (no further measures are applied)

MTFR: Maximum Technically Feasible Reduction refers to the emission levels achieved by applying all further abatement measures

Gap Closure percentage: reduction of health and environmental impacts, expressed as a percentage, of the maximum further impact reduction achievable in moving from CLE to MTFR.

4 - Uncertainties under the microscope

This section is a summary of a paper prepared by CONCAWE as a contribution to the 4th meeting of the Stakeholder Expert Group on the EU Air Policy review. The study is based on the results of extensive sensitivity analysis undertaken by CONCAWE using their in-house Integrated Assessment Model. This is largely founded on the data developed by IIASA to support their policy scenario analysis undertaken in the context of the revision of the Gothenburg Protocol (energy scenario PRIMES 2009).

The illustrative sensitivity analysis focussed on six key issues: Policy vulnerability to under-delivery of Euro VI/6 NOx emission reductions, Policy dependency on NH_3 emission reductions from Agriculture, Policy need to consider multiple time horizons, Policy vulnerability to a single energy scenario, the Policy benefit of more fully accounting for short lived climate forcers and finally, the Policy implications of differentiating the toxicity of primary and secondary components of the overall PM mix.

4.1 Uncertainty in the real world performance of Euro VI/ 6

Policy scenarios leading to revised Thematic Strategy on Air Pollution (TSAP) targets must account for uncertainties in the reductions in road transport NOx emissions associated with the introduction of Euro VI/6 standards in 2014/17.

If real-world vehicle performance results in higher than expected NOx emissions, the sensitivity analysis indicates that, at a given ambition level, this would result in significant increases in costs to the non-transport sector or even in unachievable targets.

A sensitivity analysis shows that if under real life driving conditions EURO VI only delivers a 50% improvement over Euro V and Euro 6 achieves only a Euro 5 emission level, then a factor of 3 cost increase for non-road transport sectors is possible, from 7 to 20 b€/year.

In the past, real world NOx emissions from the road transport sector have been substantially greater than forecast from the regulated emission limits (from Euro II/2 to Euro V/5), due to a significant difference between performance under actual driving conditions and performance under the standardized driving cycle that forms on which the regulation is based. This has led to substantial problems in achieving obligations under the current National Emission Ceiling Directive (NECD) and Ambien Air Quality Directive (AAQD) in a number of Member States.

The importance of this is illustrated by Figure 2 which shows the forecasted evolution in NOx emissions from Road Transport in EU-27 from 1995 out to 2030 and beyond. This is derived from CONCAWE's in-house road transport emissions forecasting model developed for and used extensively to support the European Auto Oil programmes[7].

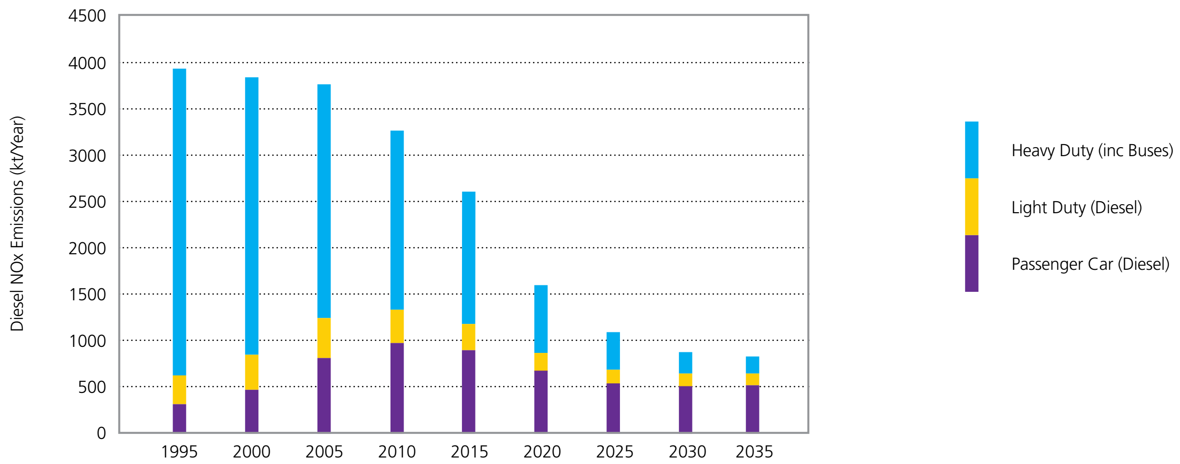
It is important to highlight the critical dependence of overall policy on the forecast transport NOx emissions. To illustrate this we compare two emissions forecasts: one based on all vehicles achieving emissions per kilometre as estimated with COPERT 4 and the other assuming higher emissions per kilometre from the Euro VI/6 diesel fleet component.

Design of sensitivity scenarios: If sensitivity scenarios are to provide insights into the influence of uncertainties on the robustness of policies they of course must have a clear basis for their design. With this in mind the following sensitivity scenarios were constructed:

Sensitivity Scenarios:
Sensitivity Scenario a: For Euro VI (heavy duty vehicles): the fleet averaged Euro VI real world NOx emission/km would be half the emissions achievable using the Euro V emission factors[8] in COPERT. **Sensitivity Scenario b: For Euro 6 (light duty vehicle):** the fleet averaged Euro 6 real world NOx emissions would be at the same level as the Euro 5 emissions represented in COPERT.

[7] The emission algorithms (e.g., COPERT 4 emission relationships) and exogenous assumptions (e.g. fleet numbers, fleet starting vintages and turnover rates) are entirely consistent with the current version of TREMOVE used to support the transport elements of GAINS.
[8] Emission factors derived from tests on marketed vehicles.

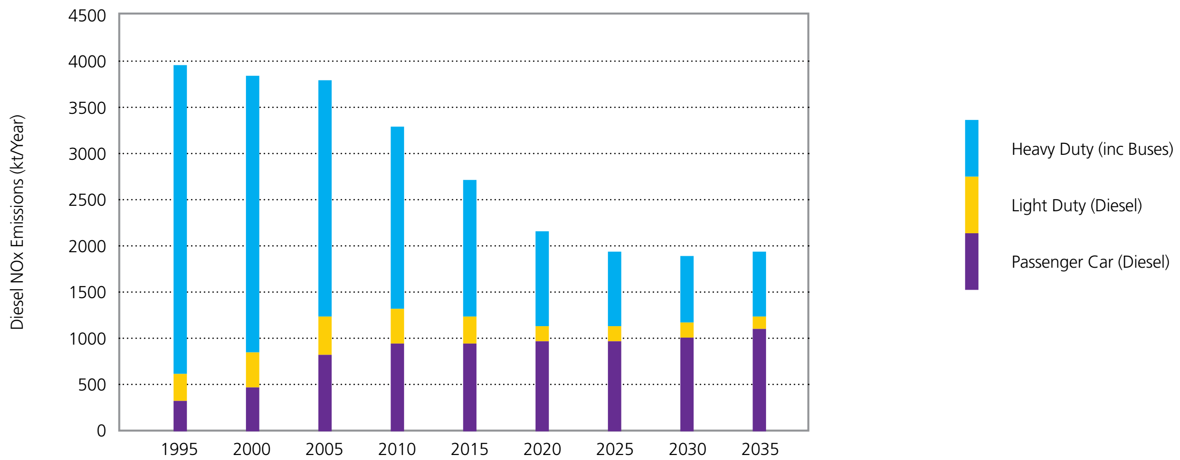
Figure 2
Evolution of NOx Emissions from Road Transport in EU-27: Base Case (COPERT 4)
Source: CONCAWE STEERS Model



Between 1995 and 2010 NOx emissions from diesel vehicles have not fallen as fast as NOx emissions from gasoline vehicles. This is in part due to growth from the dieselisation of the light duty vehicle (LDV) fleet and the general increase in vehicle kilometres driven. However, an important reason for this slower than expected reduction has been the disappointing real world performance of Euro II/2 to Euro IV/4 vehicles.

Between 2010 and 2015 with the 'real world' performance for Euro V/5 already reflected in COPERT 4, this trend is not significantly changed. In contrast by 2030 LDV diesel NOx is forecast to halve and heavy duty vehicles (HDV) NOx reduce by eightfold from the introduction of Euro 6/VI in 2015/16 when replacement of the pre 2015/16 fleet is complete.

Figure 3
Evolution of NOx Emissions from Road Transport in EU-27: Sensitivity Case.
Source: CONCAWE STEERS Model



If higher than expected emissions from Euro VI/6 vehicles do occur (sensitivity case a + sensitivity case b), NOx emissions will be double over the base case 2025 i.e. emissions would be some 1Mt/y higher. This will by far not deliver targets and may bring some Member States to a non-compliance situation.

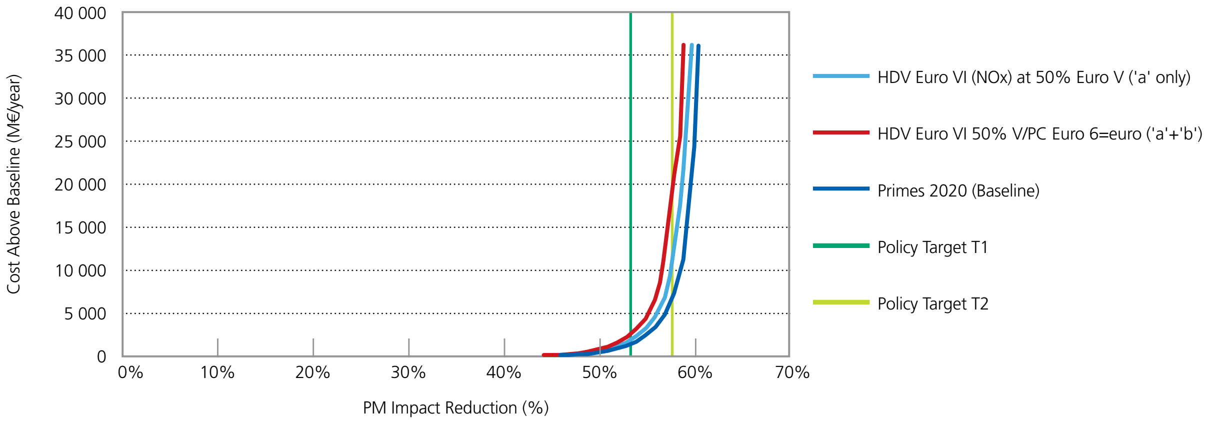
To illustrate the policy implications of this under-achievement of the Euro VI/6 program, the sensitivity case and the base case were tested under two optimisation scenarios to deliver further health impact improvement beyond the baseline (current legislation) in PM (50% gap closure[9]: Policy Target T1, 80% gap closure: Policy Target T2). The optimisations were carried out using CONCAWE's in-house Integrated Assessment Model (IAM)[10].

[9] GAP CLOSURE the reduction in impacts, expressed as a percentage, of the maximum further impact reduction achievable in moving from Current Legislation scenario to Maximum Technical Feasible Reduction.
[10] CONCAWE integrated Assessment Model utilises identical source-receptor functions, cost functions and impact algorithms to those used in GAINS to support IIASA's recent work for the revision of the Gothenburg Protocol.

The 'optimisation driver' was confined to PM health impacts to simplify the analysis and aid transparency.

Transport emissions lie outside the optimisation as they are determined by the forecast fleet development, mileage driven and technical abatement measures in place i.e. they are input data. The resulting optimised costs are for the additional stationary source abatement measures needed to achieve further PM impact reductions. Note that PM impact is related to the concentrations of total $PM_{2.5}$ in the air and this comprises both directly emitted 'primary' particles and 'secondary' particles ($PM_{2.5}$ formed in the air by chemical reaction). NOx, NH_3 and SO_2, contribute to secondary $PM_{2.5}$. The results are shown in Figure 4 below.

Figure 4
Annual Abatement EU-27 Costs For Stationary Sources (Including Agriculture).
Source CONCAWE IAM (PRIMES 2009)



Three baseline starting points were examined, all derived from the PRIMES 2009 energy scenario (central scenario for the revision of the Gothenburg Protocol).

'Base Case'- dark blue line

Energy scenario: actual baseline PRIMES 2009.

Euro VI/6: Full delivery as determined using COPERT 4 (Euro VI delivering 8 times lower emissions than Euro V and Euro 6 half of Euro 5)

With optimised delivery of a given EU-27 PM reduction target in 2020.

'Sensitivity Scenario 'a' only- light blue line

Euro VI/6: Euro VI only delivers a 50% improvement over Euro V and Euro 6 delivers as in the base case. In this case the baseline NOx emissions were adjusted in each Member State (MS) to account for the greater transport NOx emissions before the optimisation scenarios were run.

'Sensitivity Scenario 'a' and 'b'- red line

Euro VI/6: Euro VI delivers a 50% improvement over Euro V and Euro 6 performance is the same as Euro 5. Again, for this case, baseline NOx emissions were adjusted in each member state to account for the 'under-delivery' of Euro VI/6 before the optimisation scenarios were run.

The vertical lines represent two different control scenarios considered as percentage of PM impacts gap closure (GP).

Policy Target 1: 50% Gap Closure- dark green line:

Should Euro VI/6 under-deliver the implications for further investments in stationary sources (including ammonia abatement measures in agriculture) to make up for the greater than expected NOx emissions from road transport are already clearly significant. For the worst case considered in the sensitivity scenarios (sensitivity case a+b red line) Figure 4 shows annual costs doubling from some 1.5 b€/y to 3b€/y.

Policy Target 2: 80% Gap Closure- light green line:

Costs escalate since here policy would be hitting the steep part of the cost curve. In this case annual costs rise from some 7b€/y (base case, dark blue line) to almost 20b€/y (sensitivity case a+b, red line). It is also important to note that in case of under-delivery of Euro VI/6 at the higher ambition targets, in some Member States, the NOx ceilings will become unachievable even at Maximum Technically Feasible Reductions (MTFR). Such situations have already been experienced in the case of the current NECD 2010 ceilings.

It is necessary to explore the reductions that would be required from other sectors to compensate for a lower than expected delivery of Euro VI/6. Particularly in a context where the economies of the EU will increasingly struggle to compete in the global market place, these potential unintended consequences should be well understood. Certainly, the implications of such uncertainties (via sensitivity scenarios around the central policy case) need to be explored throughout the entire policy process.

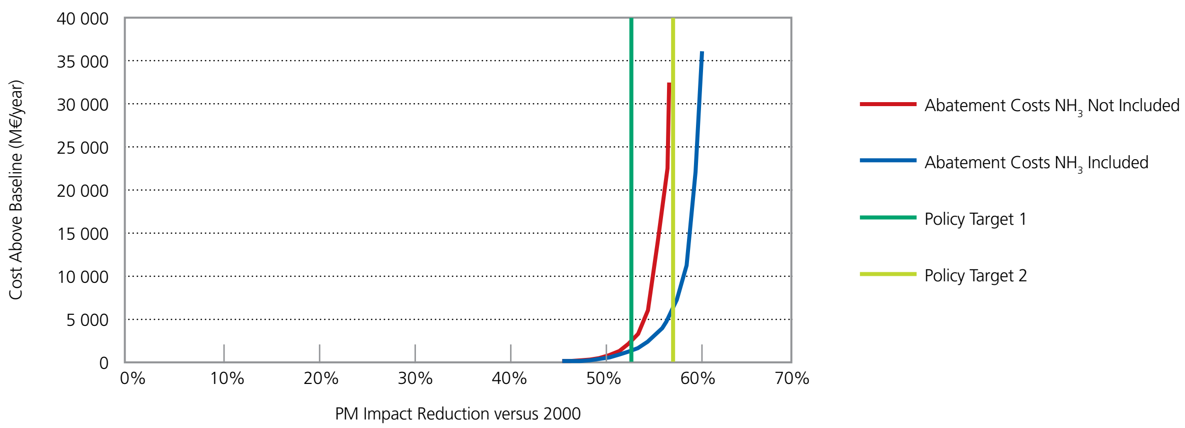
4.2 NH_3 from agriculture

Ammonia is a key pollutant; if emissions of ammonia are not reduced the scope for compensation by controls on NOx is extremely limited. It is not possible to meet ambitious acidification, eutrophication or human PM exposure targets if ammonia emissions are not reduced.

The Clean Air For Europe (CAFE) programme which underpinned the current Thematic Strategy on Air Pollution, clearly identified the reduction in ammonia emissions from agricultural sector as an important component of cost-effective policy designed to deliver improved air quality in Europe. Through earlier policy initiatives, such as the NECD and Gothenburg Protocol, the need for agriculture to be part of the solution to Eutrophication and Acidification was already well established. What was new and important in CAFE was the understanding that reductions in ammonia emissions from agriculture were central to cost-effective reductions in human exposure to fine particulates. This section illustrates why this remains crucial for any policy initiatives resulting from the review process.

CONCAWE has carried out a sensitivity analysis using its in-house integrated assessment model to identify the least-cost measures to deliver further improvements (beyond the baseline) in PM health impacts in the EU in 2020 if different NH_3 emission reduction measures are considered.

Figure 5
Annual Abatement EU-27 Costs For Stationary Sources (Including Agriculture)
Source: CONCAWE IAM, based on PRIMES 2009



Case 1- blue curve: optimised (least-cost) curve of cost versus reduction in long term health impacts of PM in the EU assuming all further abatement measures identified within the GAINS model (version used to support the GP revision work) are available for selection, including ammonia abatement measures.
Case 2- red curve: optimised (least-cost) curve of cost versus reduction in long term health impacts of PM in the EU assuming no further ammonia abatement measures are available. In other words, ammonia emissions remain at 2020 Baseline levels.

The important, even essential contribution of reductions in ammonia in achieving optimised delivery of a given PM target is evident in Figure 5.
Policy Target 1: 50% PM impact Gap closure[11]: The cost of the control scenario without ammonia abatement measures (red curve intersection with dark green line) is 3 b€/y that is essentially double of the cost with ammonia abatement measures (blue curve intersection with dark green line) that is 1.5 b€/y.
Policy Target 2: 80% PM impact Gap Closure the difference between scenarios dramatically increases from some 7 b€/y (blue curve intersection with light green line) to the Maximum Technically Feasible Reduction (MTFR) point for all the 'beyond baseline' abatement measures for stationary sources of Primary $PM_{2.5}$, NOx and SO_2 at a cost of some 32 b€/y (red curve intersection with light green line).

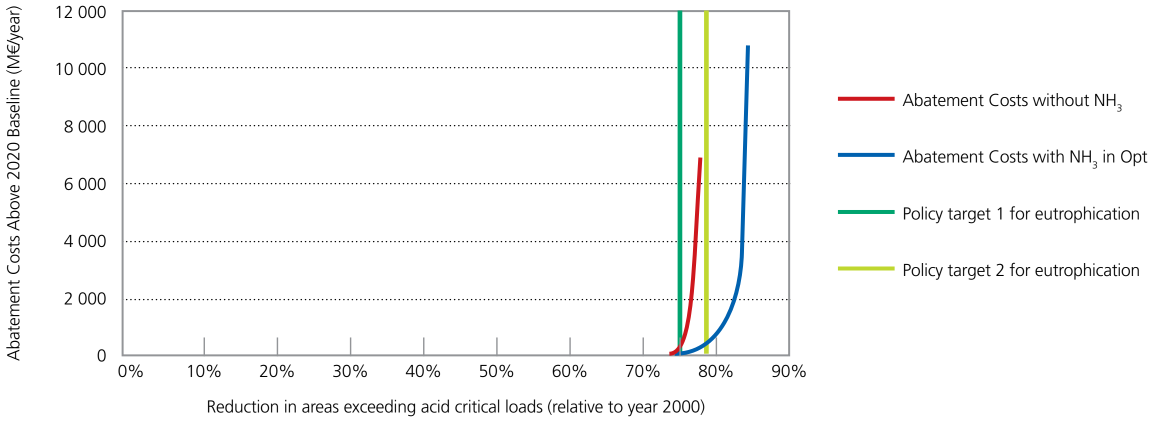
From a policy point of view, it is also worth noting that at the 7b€/y cost, the best achievable gap closure for PM[12], should ammonia emissions remain at the 2020 Baseline, is 60%. Without limit on the cost, the best achievable gap closure, as implied above, would be 80% (i.e. MTFR for SO_2, NOx and Primary PM emissions).

[11] GAP CLOSURE: the reduction in impacts, expressed as a percentage, of the maximum further impact reduction achievable in moving from Current Legislation scenario to Maximum Technical Feasible Reduction.
[12] i.e. The best achievable further health impact improvement beyond the baseline.

As already noted, ammonia reductions have long been recognised as the priority for achieving cost-effective further reductions in the areas of ecosystems exceeding acidification or eutrophication critical loads.

Figure 6 (acidification) and Figure 7 (eutrophication) show the optimised cost of further abatement measures versus reduction in the ecosystem areas exceeding their critical loads.

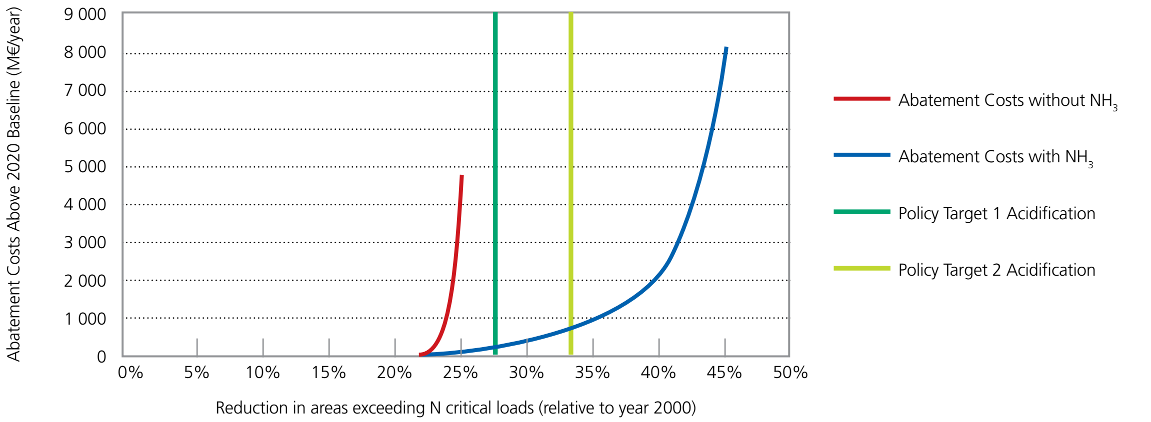
Figure 6
Annual Abatement EU-27 Costs For Stationary Sources (Including Agriculture) Above the 2020 Baseline Versus Reduction in the Area of Forest Ecosystems Exceeding Acid Critical Loads. Source: CONCAWE IAM (PRIMES 2009)



In the case of further progress in reducing acidification, the maximum further improvement is severely limited if measures are confined to SO_2 and NOx.

Case 1- blue curve: all further abatement measures are available, including ammonia abatement measures.
Case 2- red curve: ammonia emissions remain at 2020 Baseline levels.

Figure 7
Annual Abatement EU-27 Costs For Stationary Sources (Including Agriculture) Above the 2020 Baseline Versus Reduction in the Area of Ecosystems Exceeding Eutrophication Critical Loads Source: CONCAWE IAM (PRIMES 2009)



In the case of eutrophication, no significant progress can be achieved without a focus on ammonia.

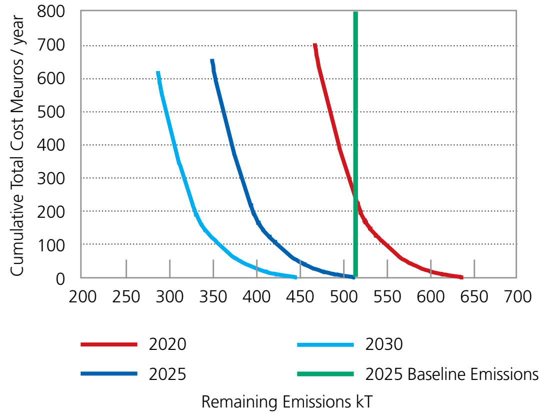
To highlight the significant challenge to the policy process of ensuring the required reductions of ammonia emissions from the agricultural sector are realised, it is worth noting, in the context of the Gothenburg Protocol (GP) that ammonia emissions in the 2020 Baseline are predicted to fall by less than 2% between now and 2020. Although a new agricultural baseline scenario is under preparation, the optimisation undertaken in this 'GP PRIMES 2009' scenario, foresees the cost-effective contribution to the 50% PM GC target to result in a 17% reduction from 'today's' level and a 29% reduction in the case of an 80% PM GC target.

4.3 Multiple time horizons

Policy horizon years are critical. The structural changes (e.g. changing energy use) and the on-going emission reductions resulting from already agreed legislation, has significant effects on emissions with time. This introduces the question of what is the appropriate timing for compliance with any new policy initiatives in a changing world. Investing heavily in abatement technology to achieve emissions reductions that will be reached by other means just a few years later could lead to unnecessary additional financial pressures and regret investment.

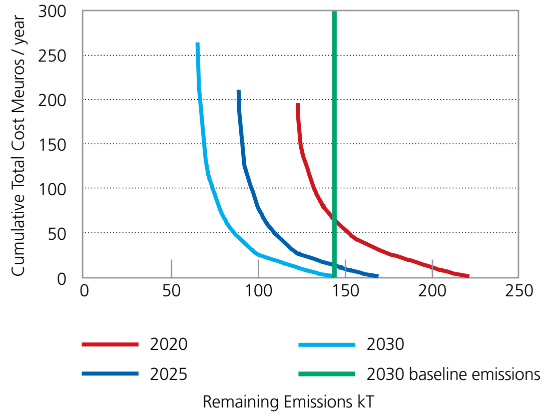
CONCAWE has carried out an analysis based on IIASA-GAINS data (IIASA report #10, (IIASA 2013)), developed for their work on the revision of the TSAP, to illustrate the economic importance of several policy horizon years.

Figure 8
National NOx Cost Curves for Three Policy Horizon Years for One Example Member State.
Source: IIASA for 2012 TSAP (PRIMES 2010)



Comparison of national cost curves for further abatement measures on NOx in one example Member State of the EU for three policy horizon years: 2020, 2025 and 2030.
In each case, the continuing effects of base case changes in emissions are clearly seen. This has significant implications for the cost of achieving further impact reductions as a function of time.
Example:
If the revised TSAP targets required this Member State to reduce its baseline NOx emission to 515kt/y, then the additional cost burden for NOx reducing measure would be:
+250M€/y, if the compliance year is 2020
+0M€/y, if the compliance year is 2025, i.e. Baseline measures achieve the objective.

Figure 9
National SO_2 Cost Curves for Three Policy Horizon Years for One Example Member State
Source: IIASA for 2012 TSAP (PRIMES 2010)



Comparison of national cost curves for further abatement measures on SO_2 in one example Member State of the EU for three policy horizon years: 2020, 2025 and 2030.
As for the NOx case, the continuing effects of base case changes in emissions are clearly seen.
Example:
If the revised TSAP targets required this Member State to reduce its baseline SO_2 emission to 144kt/y. then the additional cost burden for SO_2 reducing measure would be:
+ 60M€/y, if the compliance year is 2020
+10M€/y, if the compliance year is 2025
+0M€/y, if the compliance year is 2030, i.e. Baseline measures achieve the objective.

Of course in looking at future policies designed to make further progress in air quality in the EU it is also important to recognise the on-going costs of already agreed measures which are delivering these continued reduction in baseline emissions (with their associated further improvements in air quality) with time. For this example Member State, for NOx alone, GAINS indicates the cost of already mandated measures in 2010 to be some 2.8 b€/y, rising to 5.3 b€/y in 2020 and reaching 6.7 b€/y by 2030.

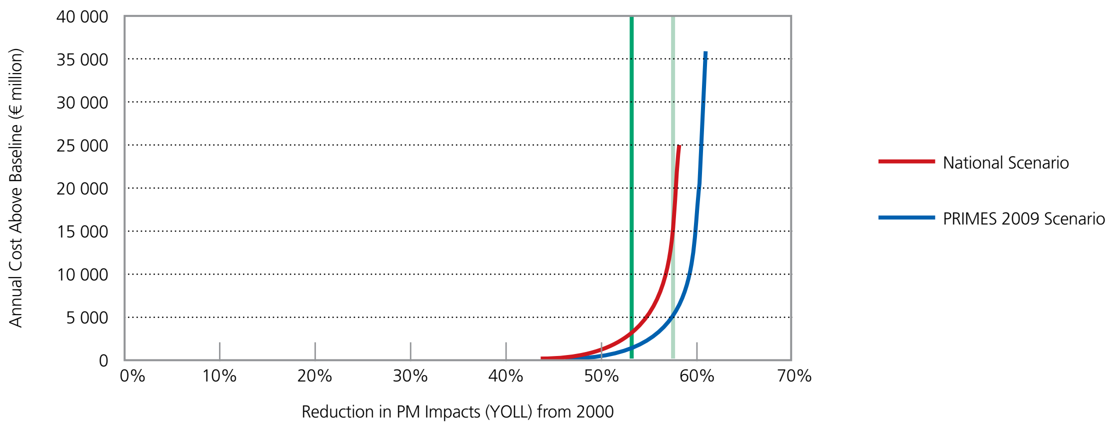
4.4. Range of Energy scenarios

Given the uncertainties in defining the 'future world' it is vital to ensure that ambition levels (expressed as revised national emission ceilings) based on one energy scenario do not result in significant escalation in compliance costs or non-achievability in a different actual future energy world. The current difficulties in some Member States in meeting 2010 NOx ceilings illustrates the vital need to include such energy uncertainties in policy development.

The need for consistency/coherency in the central assumptions used in the development of interrelated policy initiatives (e.g. Air Quality and Climate Change) is well recognised. However, this should not be interpreted as a need to base policy on a single view of the 'future world' that the policy is designed to influence. History serves as a constant reminder that actual developments can be quite different from the projections made a few years earlier. Sensitivity scenarios around a central view to test the robustness of future business plans are essential to the business world. In CONCAWE's view such sensitivity analysis is also essential in the policy arena.

In this regard, along with a number of other stakeholders, CONCAWE has requested that a range of energy scenarios, around the central PRIMES scenario, should be used in appropriate sensitivity scenarios to test policy options. In this short section, the databases used for the revision of the Gothenburg Protocol have been used to support this call.

Figure 10
Annual Abatement EU-27 Costs for Stationary Sources (Including Agriculture) Above the 2020 Baseline Versus PM Impact Reduction: Comparison of PRIMES and National Energy Scenarios Source: CONCAWE IAM



Although only twelve Member States submitted their alternative national energy scenarios during the Gothenburg Protocol review process, the consequence of moving from a PRIMES based world to this alternative 'National Energy Scenario' world is already significant. Figure 10, shows the optimised curves of cost beyond the baseline versus further reductions in PM impacts for each energy scenario. The two vertical lines indicate a medium (target 1, yellow, gap closure[13] 50%) and high (target 2 red, gap closure 75%) improvement target. The implications of arriving in the 'National energy future world' having designed policy with a sole focus on the PRIMES world are obvious: costs, justified only for the PRIMES world, double at the medium ambition level and triple to close to Maximum Technically Feasible Reduction (MTFR) costs at the high ambition. In the latter case, at an individual Member State level some individual pollutant ceilings set solely based on PRIMES would likely, at this ambition, be unachievable. Given the binding nature of the NECD, this would force Member States to consider measures that would otherwise not be justifiable and could have undesirable economic consequences. Such a situation would be avoided with the inclusion of suitable sensitivity analysis.

[13] GAP CLOSURE: the reduction in impacts, expressed as a percentage, of the maximum further impact reduction achievable in moving from Current Legislation scenario to Maximum Technical Feasible Reduction.

4.5. Short Lived Climate Forcers (SLCF)

The sensitivity scenarios in this section demonstrate how attributing a CO_2 credit or debit to SO_2, and Black Carbon emissions (based on carbon price) and including them in the optimization strategy can give an entirely different perspective to control policies and shift the policy emphasis away from NOx and SO_2 controls on stationary sources, even at relatively low carbon prices and long-time horizons.

One key recent development in the context of the revision of the Gothenburg Protocol (GP) was the inclusion of considerations over the influence of short lived climate forcers (SLCF) in the policy process with a particular focus on Black Carbon (BC). As a consequence, the GAINS team have begun to incorporate such considerations in a quantitative way into GAINS.

What this work by IIASA has provided is a helpful bringing together of quantified data on the direct global warming potential (GWP) of all the key SLCFs and was first presented by IIASA in Dublin in May 2010[14]. The following data for GWPs have been abstracted from this presentation:

Table 1
Global Warming Potentials relative to CO_2 (GWP CO_2=1) (a negative value represents a net cooling effect)

	20 year GWP	100 year GWP
SO_2	-140	-40
Black Carbon	2200	680
Organic Carbon	-240	-75

The availability of these relative GWPs allow the "CO_2 compensation costs" implied for a unit reduction in each of the three SLCF to be computed for a given carbon price e.g. the currently anticipated long-term price of €30/t CO_2e. The carbon compensation cost here is the cost involved in sustaining 'no change' in Baseline GWP by introducing compensating measures.

Table 2
Carbon compensation costs for SO_2 and BC

	Carbon compensation costs (€/tonne) Considering a carbon price of 30€/tCO_2	
	20 year integration period	100 year integration period
SO_2	4200	1200
Black Carbon	-66,000	-20,400

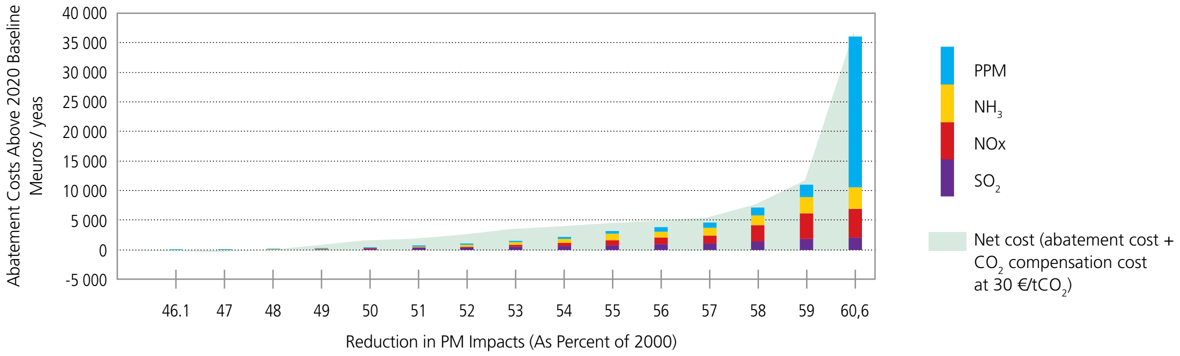
Table 2 shows that removing the beneficial climate cooling effect of sulphates derived from SO_2 emissions has to be compensated by additional climate mitigation measures. Conversely, in the case of black carbon, reductions in emissions of this powerful climate warmer result in savings in the climate mitigation costs of the baseline.

Based on detailed data made available by the GAINS team in the context of the Gothenburg Protocol revision process, CONCAWE have recently built this capability into their in-house IAM. What follows are some first results which indicate the importance of taking the full implications of SLCF into account in developing future policy. Importantly, the work clearly indicates that the inclusion of the considerations into the optimisation strategy significantly shifts the policy emphasis away from further controls for SO_2 and NOx on stationary sources, even at relatively low carbon prices and long-time horizons.

[14] First presented by Markus Amann at the 38th session of the UN-ECE TFIAM meeting in Dublin, May 17-19, 2010

Annual Abatement Costs for EU-27 by Pollutant For Stationary Sources (Including Agriculture) Above the 2020 Baseline versus PM Impact Reduction Including Carbon Compensation Cost for SLCFs

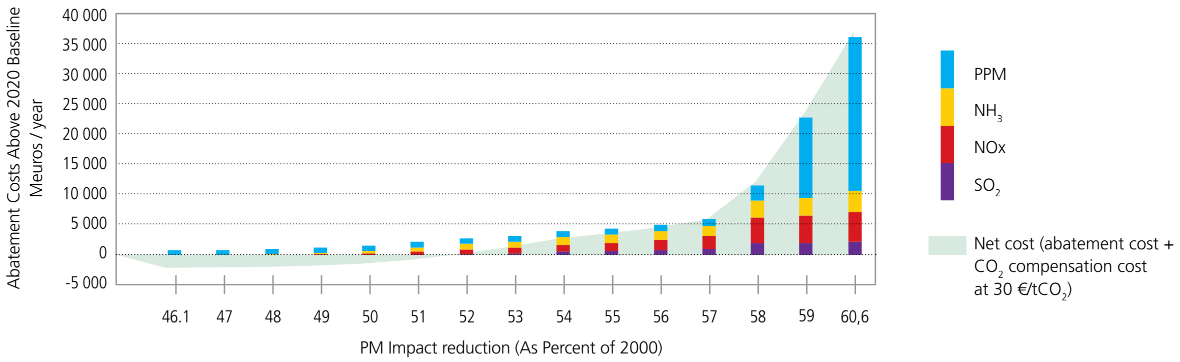
Figure 11
Case 1: PRIMES 2009 Scenario: Without SLCF in Optimisation



Case 1: Compensation cost not in the Optimisation Strategy

- Up to the 54% improvement target:
  Abatement measures on SO_2, NH_3 and NOx rather than on primary $PM_{2.5}$. As may be seen this results in a significant additional cost of measures to compensate for reductions in SLCF (here mainly SO_2).
- At the 54% improvement target point:
  Net Cost = ~1.5b€/y abatement measures + ~ 2.1b€/y CO_2 compensation cost = 3.6 b€/y
- Beyond 58% improvement target point:
  Most SO_2 measures have been exhausted and the optimiser picks Primary $PM_{2.5}$ measures. Since these emissions include a black carbon component, their reduction results in savings in the cost of climate mitigation measures included in the baseline, and the difference between abatement cost and net overall costs reduces.

Figure 12
Case 2: PRIMES 2009 Scenario: With SLCF in Optimisation



Case 2: Compensation cost not in the Optimisation Strategy

- Up to the 53% improvement target:
  Abatement measures on primary $PM_{2.5}$ with a high fraction of BC component rather than on SO_2, NH_3 and NOx.
  Net cost is negative (compensation costs > abatement cost), but abatement costs in Figure 12 themselves are clearly higher than those shown in Figure 11. In other words, as well as moving away from measures controlling secondary sources of $PM_{2.5}$, the overall abatement burden on some sectors would increase.
- Beyond 58% improvement target point:
  Most $PM_{2.5}$ measures have been exhausted and the optimiser picks SO_2, NH_3 and NOx measures.

In CONCAWE's view, these first results serve to demonstrate the importance of accounting for SLCF in the context of the current Air Policy review process as a way of properly exploiting synergies between climate change and air quality progress.

4.6. Differentiated PM toxicity

Is the assumption of 'equal toxicity' for all components of particulate matter precautionary from a Policy Perspective? Sensitivity Scenario Analysis Suggests not.

Addressing the health concerns from human exposure to fine particulates continues to be a priority concern in European air quality policy and a number of research projects have been completed in this area. Despite the recent review of evidence led by the World Health Organisation (WHO) in the project REVIHAAP (WHO, 2013), the WHO has not yet provided guidance on how to differentiate the impacts of the different components of the PM mix e.g. primary and secondary components.

As a consequence, currently all PM components are given 'equal impacts potency', under the premise that this is a precautionary assumption until the epidemiological community can provide sufficient data to support a different view.

While this continues to point to the need for more research to fill the knowledge gap, appropriately designed "uncertainty scenarios" can provide important policy input to minimise/avoid regret measures.

In all the scenario analyses carried out by the GAINS team in support of the current Air Policy review, the assumption that all components of fine particulates are equally harmful to human health has been retained. As we shall see in this section, the retention of such an assumption has profound implications for the policy outcome (e.g. a revised NECD); given that all measures to reduce PM concentrations are considered equally effective in reducing the PM impact on human health.

However, through suitably designed 'sensitivity scenarios' we can examine what the effect on policy might be if particles from some sources are more 'potent' and others less 'potent' in their effect on human health. To ensure the health impact of the overall PM mix is kept constant, if the potency of secondary particulates is reduced there is a compensating increase in the potency of primary particles.

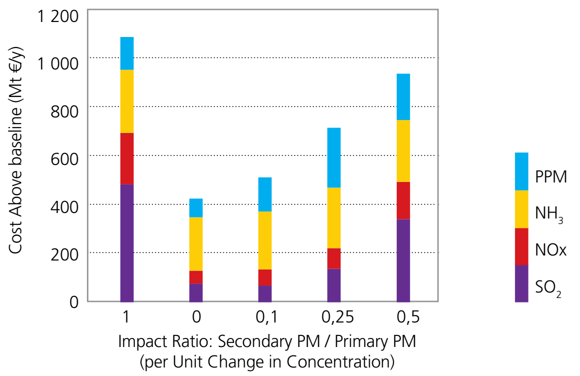
Sensitivity case- PM toxicity differentiation Primary and Secondary particles

If primary particles (derived from combustion) have more impact on human health than secondary particles, this will have implications in the control techniques selected by the integrated assessment model results because it will select emission control strategies focussed preferentially on reduction of primary particles.

- Secondary particles control: SO_2, NOx and NH_3
- Primary particle controls: particle matter (PM)

Figure 13

EU-27: Optimised Cost above Baseline (by Pollutant) to Achieve 50% Gap Closure[15] for Various Impact Ratios of Secondary/Primary PM per Unit Change in Concentration. In all cases, the overall potency of the mix is kept constant in terms of the impact on human health.



In this example, the costs of achieving a 50% PM Impact Gap Closure are shown. The bars show the additional costs for the EU-27 (expressed the annualised cost in millions of euros) above the baseline cost of CLE, based on different assumptions on the relative potency of primary and secondary particulates.

- The 100% bar shows the case where all PM, both primary and secondary, are assumed to be equally potent in their effect on human health (i.e. the assumption used in the Air Policy review). Cost above the baseline near **1.1 b€/year**.
- The 0% bar shows an extreme case where all harmful particle effects are assigned to primary $PM_{2.5}$ alone. Cost above the baseline near **0.4 b€/year**.

[15] GAP CLOSURE the reduction in impacts, expressed as a percentage, of the maximum further impact reduction achievable in moving from Current Legislation scenario to Maximum Technical Feasible Reduction.

The overall cost of mitigation measures is markedly lower in the 0% bar. This is because the potency of primary PM in this case has been substantially increased to maintain a constant overall potency of the particulate mix, so each tonne reduction has a much greater impact reduction potential. Expenditure on measures to reduce SO_2, NOx is substantially reduced; Expenditure on NH_3 is similar as a consequence of sustaining the 'come along' benefits for acidification and eutrophication achieved under the 'equal potency' scenario. The remaining three bars in Figure 13 show the effect of re-introducing the attribution of harmful effects to secondary particles.

The impact on the cost of delivering the 50% Gap closure scenario, if differentiated toxicity is assumed (especially at the low end of secondary toxicities considered) is evident from Figure 13; costs are halved. However, ensuring the right pollutants are addressed is also it is important. Table 3 shows the corresponding emission reductions by pollutant for each of the impact ratio assumptions.

This indicates the significant implications for the National Emission Ceilings Directive if a differentiated toxicity assumption were adopted.

Table 3
Emission reductions by pollutant for each impact ratio secondary/primary particles

Emission Reduction as Percent of Baseline				
Impact Ratio	**SO_2**	**NOx**	**NH_3**	**PPM**
100%	23%	6%	14%	20%
0%	7%	3%	14%	18%
10%	7%	3%	14%	20%
25%	12%	4%	14%	22%
50%	20%	5%	14%	21%

4.7. Short Lived Climate Forcers (SLCF) and PM toxicity

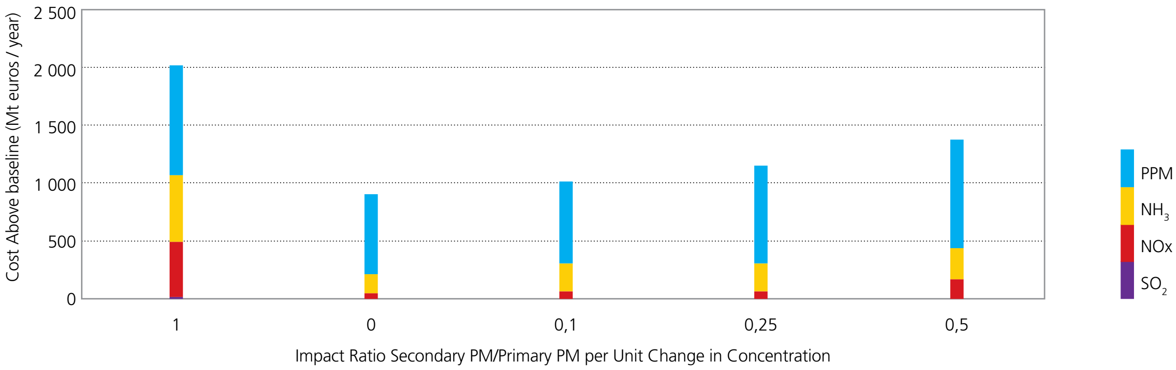
When both differentiated toxicity and SLCF are accounted for in designing an optimum policy response, even at a modest differentiated toxicity assumption, there is a profound change to the resulting package of measures and the attendant costs.

The influence on outcome of incorporating short lived climate forcers (SLCF) into the optimisation of costs for a given policy ambition was separately explored in an earlier chapter of this Review. In the scenarios depicted in Figure 13, SLCF were not incorporated in the optimisation.

To further explore the sensitivities depicted in Figure 13, further scenarios were run with SLCF inside the cost-optimisation strategy with a carbon price set at 30€/tCO_2. The results are shown in Figure 14.

Sensitivity case: Both differentiated toxicity and SLCF are accounted for in designing an optimum policy response

Figure 14
EU-27: Optimised Cost above Baseline (by Pollutant) to Achieve 50% Gap Closure[16] for Various Impact Ratios of Secondary/Primary PM per Unit Change in Concentration



Emission Reduction as Percent of Baseline

Impact Ratio	SO_2	NOx	NH_3	PPM
100%	1%	9%	19%	28%
0%	1%	2%	12%	18%
10%	0%	3%	14%	21%
25%	0%	3%	14%	25%
50%	0%	5%	14%	28%

[16] GAP CLOSURE the reduction in impacts, expressed as a percentage, of the maximum further impact reduction achievable in moving from Current Legislation scenario to Maximum Technical Feasible Reduction.

When both differentiated toxicity and SLCF are accounted for in designing an optimum policy response, even at a modest differentiated toxicity assumption, there is a profound change to the resulting package of measures and the attendant costs:

Table 4
Net costs when CO_2 compensation costs are accounted for

Impact Ratio	100%	0%	10%	25%	50%
Net cost M€	-840	-1835	-1770	-1720	-1545

The negative figures shown in Table 4 are the net costs when the CO_2 compensation costs are accounted for.

- Taking the 25% Impact ratio case in Figure 14 and comparing it to the 100% (the approach used for the Air Policy review work) case of Figure 13, starkly illustrates the extent of shift in measures/costs to deliver the policy.
- In the case of Figure 13, most money is spent on precursor emissions for secondary PM, abatement costs are some 1,100 €M/y but when CO_2 compensation costs are added, the net cost for this 50% Gap Closure essentially doubles to 2,300 €M/y.
- In contrast, Figure 14 indicates, by accounting for SLCFs and with a 25% PM impact ratio assumption, the emphasis shifts to primary PM measures, particularly those that are 'rich' in black carbon content. Given that the 'come along benefits' associated with the 'current approach' (Figure 13, 100% Impact Ratio) expenditure continues on NH_3 since this delivers Eutrophication and Acidification benefits without incurring CO_2 compensation penalties. The overall cost of abatement measures is similar but by spending on primary PM abatement and not spending on SO_2, the CO_2 'compensation' costs are negative compared to the baseline i.e. savings in CO_2 mitigation costs. Overall, this results in a saving in costs over the base case of some 1,700 €M/y compared to the 'current approach' outcome with additional costs over the baseline (including CO_2 compensation costs) of 2,300 €M/y.

5 - CBA under the microscope

CONCAWE's assessment of latest scientific information on valuation of benefits from aspects influencing mortality and morbidity factors reveals that proposed benefit numbers applied in recent Cost Benefit Analysis (CBA) studies supporting the current Air Policy review process are significantly overestimated.

Recalculating the health costs related to morbidity (in particular Reduced Activity Days (RAD) and chronic bronchitis) and mortality effects (i.e. mortality from chronic exposure to PM) of air pollution based on updated values from the scientific literature, results in a reduction of total health costs due to air pollution by a factor 4 (approximately b€ 90 per year compared to b€ 366 per year stated in IIASA Report #10 (IIASA, 2013)). Figure 15[17] provides a useful insight how a more robust calculation of chronic mortality and morbidity impacts of air pollution significantly changes the marginal benefits of air pollution control and thereby the optimal gap closure[18]. Updated health benefit figures shift the optimal gap closure to a significantly lower PM gap closure than the 75% used as central ambition level for health impacts from PM in IIASA report #10 (IIASA, 2013).

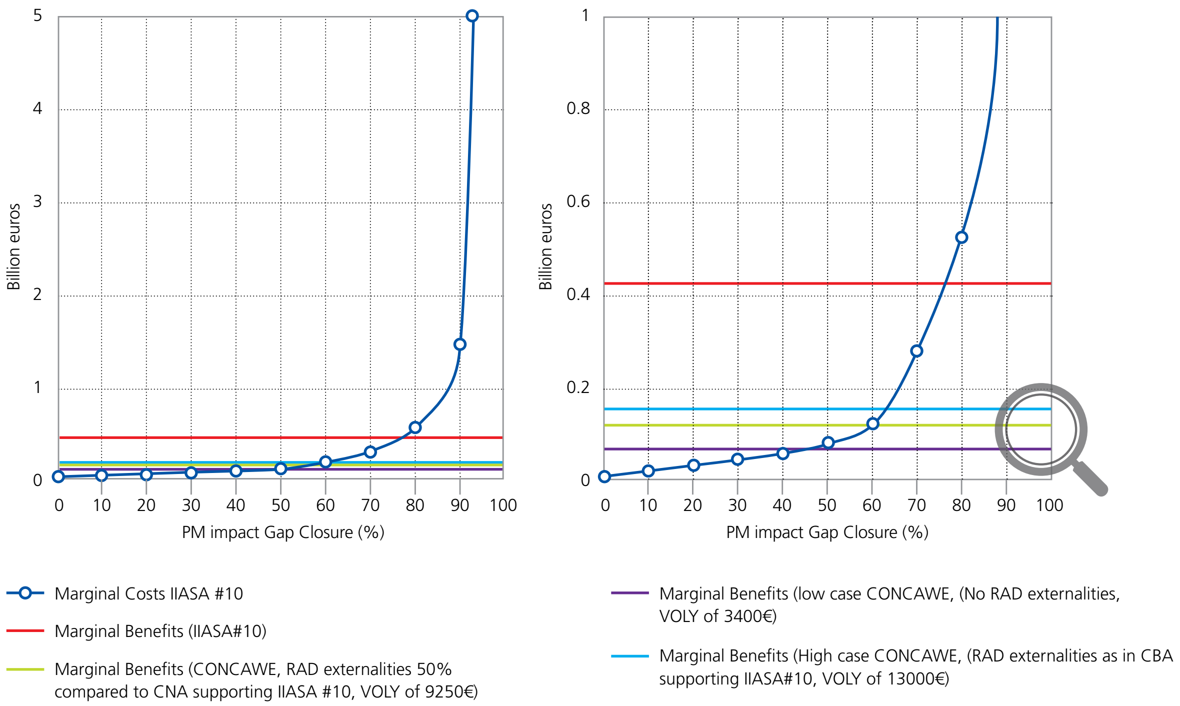
It is important to note that IIASA Report #10 (IIASA, 2013) follows a very different approach compared to the CAFE study. In the CAFE study, costs and benefits of different scenarios were compared, but in IIASA Report #10 marginal costs and benefits are compared in order to find the optimal gap closure. With this change in approach the concern is that the uncertainties in the benefit figures are very high, and the costs curves are very steep for high policy ambition levels. This means that small uncertainties in costs or benefits can significantly change the point of optimal gap closure, as demonstrated in this section.

In IIASA Report #10 (IIASA, 2013), the comparison of marginal cost and benefits excludes from the benefits all non-health related benefits, e.g., for ecosystems, agricultural crops and materials. Ecosystem services were also excluded in the CAFE study. It is important to note that including the contributions from ecosystem services do not necessarily increase the net marginal benefit. Ecosystem services are discussed in section 6 of this Review.

It is also important to recognise that generally such CBA assessments do not tell the policy maker whether the expenditure on this 'societal risk' if spent on another 'risk' would return a greater societal benefit. The need to spend proportionally across a range of policy areas is often lost in CBA.

[17] The data presented in Figure 15 are derived from digitised points taken from Figure 5.2 from IIASA TSAP report #10 (IIASA, 2013) since the source data was unavailable.
[18] GAP CLOSURE: the reduction in impacts, expressed as a percentage, of the maximum further impact reduction achievable in moving from Current Legislation scenario to Maximum Technical Feasible Reduction.

Figure 15
Comparison of marginal costs vs. marginal benefits according to IIASA calculations and CONCAWE assessments of actual health benefits. CONCAWE estimates include uncertainty assessment (low and high cases)



The justification for selecting a specific Gap Closure target is based on the monetisation of human health impacts. These are proportional to PM concentration and so the marginal benefit (the rate of change of benefit with concentration (gap closure) is a straight line. Abatement costs increase exponentially with PM reduction and so the marginal cost increases with increasing Gap Closure ambition. This figure shows on two scales how the valuation given to health impacts drives the policy ambition which should not go past the point where the marginal cost of measures equals the marginal benefits.

The sensitivity interval for the gap closure ambition ranges from 45% (low case CONCAWE) to 63% (high case CONCAWE) based on elements around valuation of health benefits in a CBA. The variables in this sensitivity analysis are twofold: a) the VOLY value applied (€ 13,000, € 9,250 or € 3400) and b) the value of Restricted Activity Days (RAD) reduction compared to IIASA Report #10 (ranging from unchanged to total exclusion due to attached major uncertainties; (IIASA, 2013)). This more robust valuation of health benefits results from a consequent application of latest scientific health data from peer-reviewed literature, most applicable methodology to determine a VOLY value for the valuation of chronic mortality impacts, and more robust valuation of morbidity (Chronic bronchitis and RADs) and chronic mortality impacts.

Table 5
Comparison of mortality and morbidity benefits stated in IIASA report 10 and CONCAWE recalculation

	IIASA report #10 data billion€/year, 2005 prices	CONCAWE estimation billion€/year, 2005 prices
Mortality from Chronic Exposure to PM	261	42
Chronic Bronchitis	42	6
Reduced Activity Days	35	17
Other Morbidity benefits	28	28[19]
Total	**366**	**93**

[19] CONCAWE did not recalculate "other morbidity costs", the study focused on Reduced Activity Days, chronic bronchitis and mortality from chronic exposure to PM.

Cost-benefit analysis (CBA) is increasingly referred to as a basis to support target setting for air quality policies. With CBA the societal costs and benefits of different ambition levels can be compared, provided that both costs and benefits are expressed in a monetary unit. Recent CBAs (AEA, 2011; EC4MACS, 2011; EEA, 2011a) conducted in support of European air quality policies have focused on comparing costs and benefits, each comprising a mix of targets for reducing the ambient concentrations of PM, ozone, acidifying and eutrophying substances. The benefits are mainly driven by the particular value given to the statistical improvements in average life expectancy arising from reduced exposure to fine particulates.

Since CBA is having an increasing role in the target setting process of the current Air Policy review, it is crucial that it is applied in a scientifically robust manner. However, CONCAWE sees that there are at present several important flaws and limitations in the way CBA is applied in the Air Policy review process.

This section of the special issue summarises a set of CONCAWE Review articles, CONCAWE submissions to the Stakeholder Expert Group (SEG) engagement process and some more recent assessments by CONCAWE to address the difficulties and improvement opportunities in evaluating health effects in environmental CBAs.

The limitations, in the CBAs cited above, are specific methodological inaccuracies regarding the valuation of the benefits associated with improvement of human health, and uncertainties of scientific data underlying the valuation process and conceptual limitations in the interpretation of CBA outcomes for policy-making:

(i) Basics principles for valuating health impacts in a cost-benefit analysis
(ii) Values for monetising chronic mortality effects
(iii) Values for monetising morbidity effects
(iv) Quality and uncertainty of morbidity health effects of air pollution
(v) Insufficient uncertainty analysis to analyse the repercussions on the costs and benefits of different policy targets.
(vi) Fundamental limitations of applying CBA for policy formulation

Figure 15. and the box above summarise the key findings of CONCAWE's assessments, while the sections 5.1 to 5.6 provide more of the rationale behind these outcomes.

5.1. Basics for evaluating health impacts in an environmental cost benefit analysis

Estimating the monetary benefits to society of health improvements is a complex endeavour. To start with, it is essential to select the correct metric. In the context of air pollution CONCAWE strongly believes that VOLY is the only appropriate metric to assess chronic mortality effects caused by air pollution, where health effects are hugely dominated by PM.

It is a complex exercise to assign a monetary value to changes in human health impacts due to air pollution. For the first time this has been done in the Clean Air For Europe (CAFE) Cost Benefit Analysis (CBA). Two aspects are of specific relevance:

- The choice of the right metric (or 'unit of measurement') to express the health impacts that will be discussed in this section,
- The monetary valuation of this metric, as outlined in section 5.2.

Two concepts are often used to assign a monetary value to changes in human mortality. A metric that is often used is called the Value of a Statistical Life (VSL) or the Value of a Prevented Fatality (VPF). The VPF is the amount of money that a community of people is willing to pay to lower the risk of one anonymous instantaneous premature death within that community (e.g. by certain traffic safety measures). Whereas to save a specific individual in danger usually no means are spared the VPF is about lowering the risk of premature death in the statistical sense. This leads to a finite value for VPF. VPF is the correct metric within a context of observable deaths, e.g. in traffic accidents.

However, in the context of air pollution, as set forth in the CAFE methodology (2005), the health impact especially of particulate matter (PM), can be described much more adequately in terms of a shortening of the life expectancy of people (often called chronic mortality) rather than by attributable deaths.

"Consistent with WHO guidance, our own established practice, and a wider emerging consensus in favour of using life table methods, the analysis will express health impacts in terms of years of life lost from air pollution. The study team also recommends years of life lost as the most relevant metric for valuation..."

AEA, 2005

Therefore, the so called Value of a Life Year (VOLY) is the only appropriate metric. The VOLY is the amount of money associated with an increase in statistical life expectancy of one year.

More details, including all references, can be found in CONCAWE 2006a and CONCAWE 2012b as well as in a CONCAWE report (CONCAWE, 2006b).

5.2. More robust monetisation of Chronic Mortality impacts

CONCAWE proposes to use a more robust way to calculate a "Value of a Life Year" (VOLY) from a given Willingness to Pay (WTP) survey, this is the so called "Maximising Societal Revenue" (MSR) that respects the WTP choices of the whole population included in the survey. CONCAWE believes that the data of the more advanced WTP studies should be used to derive the VOLY value and has done so in this assessment. As a consequence, the monetised benefits for chronic mortality aspects would decrease by a factor of 6.2 (VOLY value of NewExt[20] study adjusted to 57,700 € vs. 9250 € of CONCAWE's MSR value).

The effect of long-term exposure to fine particulate matter has emerged as the most important health issue resulting in reduction in life-expectancy. The monetisation or valuation of mortality impacts of air pollution, and therefore of the benefits of its reduction, is carried out by calculating the so called "Value Of a Life Year" (VOLY), which is the amount of money associated with an increased statistical life expectancy of one year. VOLYs are generally derived via execution of Willingness to Pay (WTP) surveys. In such WTP surveys people are interviewed for their WTP to achieve a small increase in statistical life expectancy (see CONCAWE, 2012b).

The method followed in the cost-benefit analysis used in the Air Policy review presents a set of issues summarised in the boxes below.

The different valuation of the mortality impacts in the benefits calculation of air pollution reduction measures. This is because the mortality aspect (i.e. reduced statistical life expectancy from exposure to PM) driven by the CAFE[21] VOLY value represents around 70 to 75% of the total health benefits and hence dominates the benefits associated with CBA. That is why it is important to apply the most robust methodology to determine a reliable / realistic VOLY value. In CONCAWE's view the MSR method combined with the data derived from more advanced WTP studies reflects current best practice. CONCAWE supports the weighted average VOLY range from €3,400 to €13,000 with a mid VOLY value of €9.250 (all values are not corrected for inflation; see section D in the box below). Consequently, the benefits for mortality aspects would decrease by a factor of 6.2 (VOLY value of NewExt study 57,700 € (NewExt, 2004) vs. 9250 € of CONCAWE's MSR value; the NewExt values are still being applied in the current Air Policy review process) and the overall health benefits by a factor of 4. The more robust valuation of morbidity impacts is discussed in Section 5.3 and 5.4, which represents the biggest part (20 to 25%) of the remaining overall health benefits.

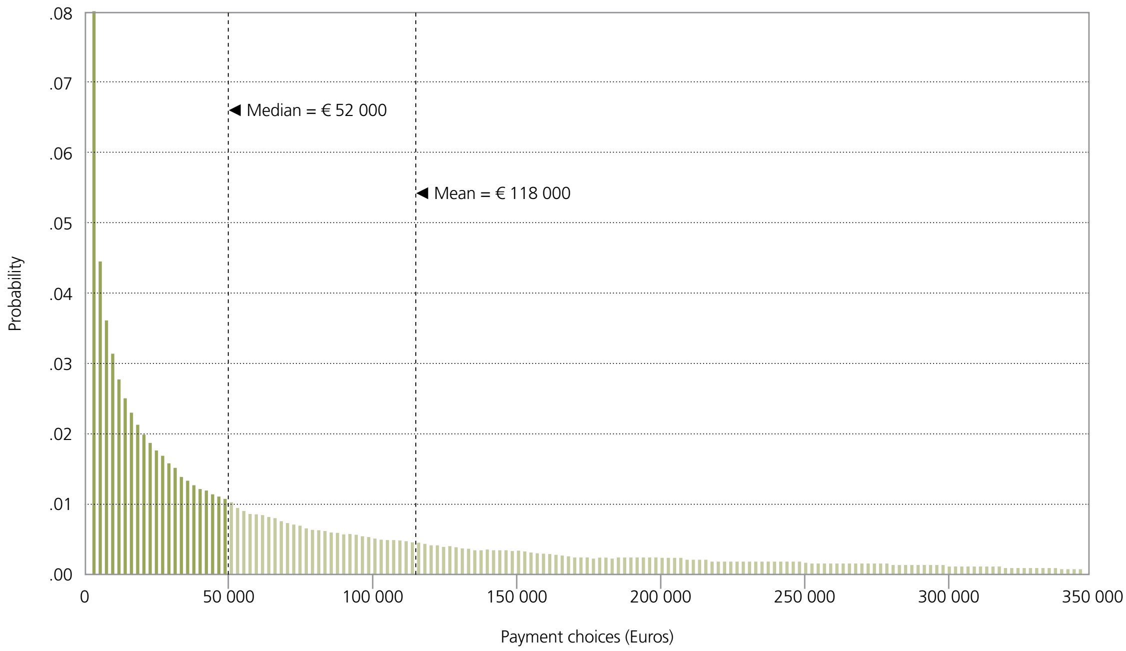
[20] The NewExt study assessed external costs of energy technologies. Its findings were used in the Cost Benefit Analysis supporting the CAFE program during the previous revision of air policy in Europe (NewExt,2004)
[21] CAFE: Clean Air For Europe programme 2005.

Issues Deriving the Value Of Life Year (VOLY)

A) Variance in Willingness to Pay choices:

- The responses from interviewed participants represent 'virtual' rather than 'real' money,
- The responses are very varied and particular highly skewed at one end (towards zero) (Figure 16).
- Formulation of survey questions has an important effect on made choices:
  - In particular the responses (scaled up to a year) are not proportional to the risk reduction, but depend on the size of the risk reduction assessed; it appears that longer life expectancy is disfavoured relatively speaking as it can be seen in Figure 17 for the results of the subsets of the NEEDS study, where WTP for 3 and 6 months increase in life-expectancy were elicited (Desaigues et al. 2007 & 2011) and the DEFRA study where WTP for 1, 3 and 6 months increase in life-expectancy were elicited (Chilton et al. 2004).
  - If it is suggested respondents will not be in good health then lower values are obtained.

Figure 16
Highly skewed distribution of WTP values as forecasted from the Weibull distribution parameters of the NewExt survey in 2004 (CONCAWE, 2006a), adjusted for inflation and used as a basis for the 57,700€ value used in the Air Policy review CBA



B) Variance among Willingness to Pay studies:

- Derived VOLY values are very different across a number of elicited WTP studies (Figure 17).

C) Challenges in determination of the most appropriate VOLY value:

- The CBA community acknowledged during the Clean Air For Europe (CAFE) program that the most representative VOLY could be obtained by statistical analysis using the full distribution range instead of median or mean values as these do not respect individual WTP choices; it just marks the dividing price for the risk reduction where 50% say they are not willing to pay more for the statistical benefit. However policy-makers prefer a single reference VOLY (median or mean) for ease of communication.
- Assessing the available WTP surveys (Figure 17), the NewExt study is the least appropriate WTP study due to the following restrictions:
  - NewExt was designed to develop data for prevented fatality (VPF); to derive a VOLY value an inappropriate methodology was applied by back-calculating from VPF/VOSL.
  - Its inappropriateness is evident considering NewExt results deviate strongly from those of other WTP studies (Figure 17; Table 6).

CONCAWE strongly believes that the VOLY values derived from the NewExt are inappropriate and should not be considered in CBA analyses for the ongoing or any future air policy rounds. Instead CONCAWE proposes to use an advanced methodology, which is briefly summarised below (see section D in this box).

However, all recent CBAs (AEA, 2011; EC4MACS, 2011; EEA, 2011a) supporting the Air Policy reviews continue to use the median VOLY value of the NewExt study (2004). The CBAs simply adjusted its values for inflation in the geographical zone of the individual study (inflation corrector differs as a function of different geographical scope of the studies because of different inflation rates for each country), despite better insights from more recent studies.

D) Advanced methodology "Maximising Societal Revenue" to derive a robust VOLY value

- CONCAWE proposes to use a more robust way to calculate a VOLY from a given Willingness to Pay (WTP) survey, which respects individual expressions of WTP of all the individuals surveyed.
- This is achieved by a simple flat fee analysis to determine VOLY from a WTP survey, it assumes a pay/no pay threshold and sets the threshold (fee) to maximise the sum that can be raised from the survey population. This is a technically and methodologically more robust approach compared to using median or mean values of a survey. The biggest advantage is that this flat fee approach reflects the full distribution of expressed WTP values and is less sensitive to the very highest and lowest choices.
- How the MSR values relate to the median and mean values is shown in Table 6 and Figure 17 for a number of elicited WTP studies. It is worth noting that the MSR values correspond more closely to the median and are far away from the mean VOLY values of each of the surveys.
- More detail can be found in (CONCAWE 2012a).

Figure 17
The (forecasted) VOLY distributions (20 to 80 percentiles) according to three studies, also indicating the location of the median, mean and MSR value

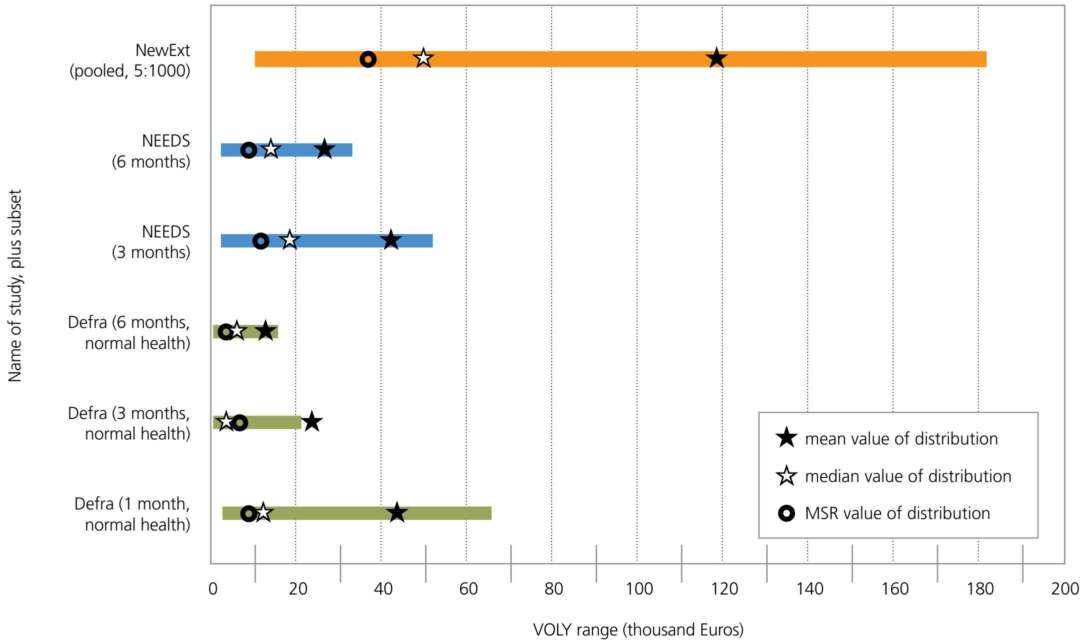


Table 6
Derived VOLY (€ per statistical life year lost) of more advanced WTP studies (no correction for inflation applied)

WTP study	VOLY Median	VOLY Mean	VOLY based on MSR[22]
NEEDS - 6 months[23]	14,000	27,000	9,100
NEEDS - 3 months[23]	19,000	42,000	13,000
DEFRA - 6 month[24]	2,700	13,000	3,400
DEFRA - 3 months[24]	2,200	23,000	5,500
DEFRA - 1 month[24]	15,000	45,000	13,000
Weighted average VOLY of all studies	**11,600**	**31,000**	**9,250**

[22] CONCAWE, 2012a.
[23] Chilton et al. 2004.
[24] Desaigues et al., 2007. Note that '1 month', '3 months' and '6 months' refer to the different risk-reduction choices in these WTP studies. The values represent averages of assessments for normal health for each risk-reduction choice.

5.3. More robust monetisation of Morbidity impacts

The results of the recent studies indicate that the value range of chronic bronchitis is approximately 7 times lower than considered in the CBA studies. And as the uncertainties around monetising Restricted Activity Days (RAD) are substantial, CONCAWE has strong reservations about the monetisation of RAD. Therefore in CONCAWE's valuation it was assumed as base case that the health cost of RAD would be halved, representing a more conservative estimate which is in line with the only recent study on RAD by Maca et al. (2011). In this more realistic valuation, the benefits associated with morbidity are halved.

The morbidity impact is also valued and considered as part of the total economic benefits of the improvement in the air quality. The costs of morbidity typically account for 20-25% of the costs of air pollution in Europe as calculated for the Air Policy review but will be proportionally higher if the cost of chronic mortality is given a more realistic value (see Section 5.2). Hence, it is important to ensure that the morbidity impacts and the monetary values of these effects are robust

The morbidity impacts of air pollution are calculated from two main components: the new chronic bronchitis cases and the restricted activity days (RADs).

- Valuation of chronic bronchitis: this is done by taking into account the costs of medication and the willingness to pay studies to avoid the symptoms of chronic bronchitis.
- Valuation of Restricted Activity Days: this is done by taking into account the results of the willingness to pay studies to avoid a day of restricted activity.

CONCAWE has analysed the various approaches used in the scientific literature to value the costs of chronic bronchitis and RADs and how these values are included in recent CBAs considered in the development of European air policies. The assessment shows that there is an urgent need to reconsider the cost values currently attributed to morbidity effects in the European CBAs on air pollution control.

Issues Deriving the Morbidity Impacts

Values of chronic bronchitis and RADs are based on very few valuation studies using the Willingness-to-Pay (WTP) principle (see also Section 5.2), based on limited sample surveys, and with inconsistent results for some of the studies.

Chronic Bronchitis

Based on an analysis by CONCAWE of recent studies available in the literature[25], the analysis reveals a more up-to-date value range for monetising new cases of chronic bronchitis from €25,000 to €28,000 per case rather than the €208,000 currently applied in CBAs (e.g. IIASA GAINS model, EEA 2011a). The complete analysis, which was also made available during the SEG process, is available in electronic form in CONCAWE website[26]. CBA under the Microscope of this issue.

The results of the recent studies indicate that the value range of chronic bronchitis is approximately 7 times lower than considered in the CBA studies. If this new values were considered the benefits associated with morbidity would decrease by a factor of 1.5.

Restricted Activity Days (RADs)

There are important uncertainties in relation to the assumed monetised value used in the CBA studies.

1. The value is based upon just one study (Ready et al., 2004)
2. The results of this one study (Ready et al., 2004) are internally inconsistent. Specifically the WTP of Spanish and Portuguese to avoid a RAD far exceeds that of northern Europeans, and the WTP to avoid a minor RAD exceeds the WTP to avoid a 'regular' RAD in Spain. This may be related to an incorrect application of the survey.

A more recent and thorough study (Maca et al., 2011), presents estimates for the costs of RAD that are around a factor 2 lower than the value by Ready et al, 2004.

Considering these uncertainties CONCAWE has strong reservations about the monetisation of RAD. Therefore, in CONCAWE's valuation it was assumed as base case that the health cost of RAD would be halved, representing a more conservative estimate which is in line with the only recent study on RAD by Maca et al. (2011).

[25] Menn et al., 2012; Maca et al., 2011; Chapman et al., 2006; Stavem, 2002; Wilson et al., 2000; Priez and Jeanrenaud, 1999; O'Conor and Blomquist, 1997

[26] The original contribution to the Stakeholder consultation "CBA under the Microscope" is included in the electronic version of this Special Review which can be found on CONCAWE website www.concawe.eu.

5.4. Quality and uncertainty of morbidity health effects of air pollution

CONCAWE has not only concerns regarding the monetisation of the health effects of air pollution, but also on the soundness of certain underlying morbidity health data.

Studies on chronic bronchitis and Restricted Activity Days (RADs) used to quantify morbidity health impacts of air pollution have severe shortcomings that should limit their application in the policy formulation process.

In CONCAWE's view, the studies proposed for use in the CBA, Abbey et al. (1995) for bronchitis, and Ostro et al. (1987) and Ostro and Rothchild (1989) for RADs, are not of sufficient quality for use in a CBA. CONCAWE's scientific concerns with these studies are summarised in the note on Concentration-Response Functions for Morbidity Endpoints under the Project HRAPIE)[27]. The complete note can be found in the report in the electronic form on CONCAWE website.

Chronic Bronchitis study (Abbey et al. 1995) – Summary of CONCAWE concerns:

- Use of the imprecise exposure metric of PM_{10} estimates derive from measurement of total suspended particulates and airport visibility data.
- PM_{10} risk estimates have been converted to $PM_{2.5}$ estimates for purposes of the CBA. Since bronchitis is primarily a disease of the upper respiratory tract, it is inappropriate to attribute bronchitis to $PM_{2.5}$, a pollutant that distributes in the lower respiratory tract. In CONCAWE's view the exposure response functions (ERFs) are inflated as they are based on high levels of air pollution in California 20-30 years ago, and hence are not applicable to evaluation of air pollution in Europe today. Results of the study are confounded by lack of control for smoking, a well-known risk factor for development of bronchitis. In our view, a single study reporting a non-statistically significant result should not be used in a CBA.

Reduced Activity Days studies, RAD, (Ostro et al. (1987) and Ostro and Rothchild (1989)) – Summary of CONCAWE concerns

- $PM_{2.5}$ levels were not measured. Rather, $PM_{2.5}$ levels were estimated from PM_{10} measurements and visibility data from airports.
- The air pollution data evaluated were high levels in existence in California over 30 years ago are not applicable to Europe today.
- The health endpoint of RAD is highly subject to socioeconomic confounding. Significant city to city differences in RAD rates were observed in the studies used to derive the exposure response factors.

This was likely due to socioeconomic factors and other factors that were not adequately controlled in the selected studies:

- Time spent outdoors, building construction, health practices including how such days are recorded, age of the population, sex, race, education, income, marital status, temperature, employment conditions and rates, smoking rates, and many other factors. Even greater differences would be expected when extrapolating the results of these studies for use in Europe.

[27] CONCAWE comments on Concentration-Response Functions for Morbidity Endpoints under the Project HRAPIE can be found as an original contributions to the Stakeholder consultation in the electronic version of this Special Review which can be found on CONCAWE website www.concawe.eu

5.5. Appropriate uncertainty analysis on CBA

There are considerable uncertainties accompanied with CBA for the air policy development. The understanding of the extent of and the effect of these uncertainties is insufficient. Relatively small changes in each source of uncertainty could have significant effect on the cost-benefit ratio of different emission control strategies. A better understanding of these uncertainties and/or careful consideration of the uncertainties is required to achieve a robust (revision of the) air policy package.

At present there is insufficient understanding of the effects of the various sources of uncertainty on the outcomes of the CBAs. These uncertainties pertain to:

- Future baseline emissions as a function of economic developments may affect the marginal cost/benefit ratio of different air policy ambition levels or may lead to technically unattainable ambition levels. (see also Section 4)
- Future energy prices and impacts on future emissions can affect the marginal cost/benefit ratio of air pollution control ambition levels. (see also Section 4)
- Future ambient concentrations as a function of emission reductions and other factors (including weather patterns and wind directions that may be influenced by climate change)
- Exposure levels of people as a function of human behaviour
- Monetary valuation of health and environmental effects.

Relatively minor changes in each of these factors could have significant repercussions for the cost-benefit ratio of different emission control strategies. In order to ensure the selection of no-regret policy measures, there needs to be an understanding of how robust the cost and benefit assessments are prior to deriving any ambition level in the air policy.

It is also relevant to note that care needs to be taken in applying singular CBAs as a basis for policy-setting (See section 5.6).

A proper uncertainty analysis of the valuation of health benefit needs to consider a range of valuation studies including a sensitivity analysis of critical assumptions. This approach has not been taken in recent CBA studies (AEA, 2011; EC4MACS, 2011; EEA, 2011a) undertaken in support of the Air Policy review process. Moreover, these CBAs did not account for the more recently available studies in the field of mortality and morbidity valuation.

Besides uncertainties related to the methodology and execution of CBAs, this special issue of a CONCAWE Review addresses also other uncertainties that refer to aspects beyond or outside CBA, such as anticipated energy scenarios, multiple time horizons for policy formulation (see Section 4).

5.6. Fundamental limitations to keep in mind when using CBA for policy formulation

In addition to the methodological inaccuracies in current CBAs (see sections 5.1, 5.2, 5.3, 5.4 and 5.5), there are two aspects that need to be kept in mind when interpreting and using CBA outcomes in the context of EU air policy:

- Firstly, care needs to be taken in applying singular CBAs as a basis for policy-setting. To ensure that policy development is robust, it is important that the policy does not focus on a single issue/value.
  - Emission control measures have been implemented over many decades to successfully reduce national pollutant emissions. Taking further reduction actions will soon result in diminishing returns and escalating costs. This implies that there may be more cost-effective ways to achieve certain health or environmental benefits compared to reducing air pollution.
  - Integration of Climate and Air Policy measures: include the short lived climate forcers effect as a factor in the cost-benefit analysis (see also Section 4.5).

- Secondly, only health benefits and ozone damage to some crops are quantified in the monetary analyses. The costs and benefits of eutrophication and acidification control are not quantified and therefore not included in the CBA models constructed to date (AEA, 2011; EC4MACS, 2011; EEA, 2011a). Acidification and eutrophication affect ecosystems and the benefits provided by ecosystems to people ('ecosystem services', see Section 6), and eutrophication may lead to increases in some ecosystem services (for instance timber production in nitrogen limited forests) and decreases in the supply of other services (for instance biodiversity). However, it is important to note that including these contributions do not necessarily increase the net marginal benefit.

  Even if these costs and benefits are not quantified, ambitious targets for eutrophication and acidification control are included in the policy scenarios for air pollution control (A1-A6) presented in IIASA Report #10 (IIASA, 2013). The ambition levels for eutrophication and acidification have been established by association with the anticipated health benefits following PM emission reduction instead of on the basis of an analysis of the costs and benefits associated with the reduction of eutrophication and acidification themselves. In order for CBA to properly play its role in informing the ambition setting process for meeting multiple targets in the most cost effective way (and provide transparency in the final impact assessment) it needs to be in a position to correctly attribute the incremental benefits and associated incremental costs for meeting each individual target. This is vital to ensure that benefits derived from achieving one target (e.g. PM health impact reduction) are not used to 'subsidise' the limited monetised benefits or lack of monetised benefits for meeting the additional target(s) (e.g. ozone health impact reduction). Therefore, eutrophication and acidification targets should be subject to a specific incremental CBA rather than be included in policy scenario's where the monetary benefits are driven by health impacts. Each of these two aspects needs further consideration and evaluation/assessment in the near future in order to allow a more robust policy formulation. Until a better resolution is achieved, these aspects limit the applicability of CBAs.

Some more information is available in the original CONCAWE contribution to the Air Policy review Stakeholder consultation process "CBA under the Microscope".

6 - Ecosystem services under the microscope

Ecosystem approaches can be defined as 'approaches to environmental management and policy making that aim to compare costs and benefits of management and policy options on the basis of an analysis of their impacts on the supply of benefits from ecosystems to people'.

The benefits supplied by ecosystems to people have been labelled 'ecosystem services' (MA, 2003; TEEB, 2010) and comprise such benefits as the provisioning of goods by ecosystems (e.g. wood, fish, genetic information), the regulation of environmental processes (e.g. water purification by wetlands, carbon sequestration in forests) and cultural services supplied by ecosystems (e.g. providing opportunities for recreation). The capacities of ecosystems to supply such services can be affected by air pollution and other types of environmental stress. Ecosystem degradation, for instance through air pollution, evokes a cost expressed through a reduced supply of ecosystem services, and ecosystem rehabilitation through reduction of air pollution may lead to economic benefits through enhanced ecosystem services supply.

This section focuses on ecosystem approaches applied to the field of air emissions. The potential impact of air pollution is short as well as long-range and, through reaction and transformation in the atmosphere, anthropogenic and biogenic emissions can combine to create pollutants having adverse effects on ecosystems and human health. Section 6.1 provides an overview of ecosystem approaches to environmental management. Section 6.2 examines how ecosystem approaches are being used in support of two on-going European policy formulation processes, the Gothenburg Protocol and TSAP reviews. Section 6.3 analyses methodological gaps and uncertainties and proposes means of resolving these. More details on this section can be found in Appendix 1: Ecosystem services under the microscope.

6.1. Key concepts

6.1.1. Ecosystem services

Definition:
Ecosystem services are the goods or services provided by the ecosystem to society (MA, 2003). The supply of ecosystem services will often be variable over time, and both actual and potential future supplies of services should be included in the consideration of ecosystem services in support of environmental policy making. Ecosystems include fully natural systems, but in recent thinking also systems that are strongly modified or influenced by people, such as croplands[28].

In an Ecosystem Approach, the state or health of the ecosystem is linked to the supply of ecosystem services. Changes in ecosystem state will commonly affect ecosystem services supply, with different ecosystem services affected in a different manner.

Classification of Ecosystem services
There are three different categories of ecosystem services that are generally distinguished (MA, 2003; TEEB, 2010; EEA, 2011b): (i) provisioning services; (ii) regulation services; and (iii) cultural services. These categories are described below.

(i) Provisioning services	Reflect goods and services produced by or in the ecosystem, for example a piece of fruit or a plant with pharmaceutical properties. The goods and services may be provided by natural, semi-natural and agricultural systems and, in the calculation of the value of the service, the relevant production and harvest costs have to be considered.
(ii) Regulating services	Result from the capacity of ecosystems to regulate climate, hydrological and bio-chemical cycles, earth surface processes, and a variety of biological processes. These services often have an important spatial aspect; e.g. the flood control service of an upper watershed forest is only relevant in the flood zone downstream of the forest.
(iii) Cultural services	Relate to the non-material benefits people obtain from ecosystems through recreation, cognitive development, relaxation, and spiritual reflection. This may involve actual visits to the area, indirectly enjoying the ecosystem (e.g. through nature movies), or gaining satisfaction from the knowledge that an ecosystem containing important biodiversity or cultural monuments will be preserved.

[28] Key publications in the field are the Millennium Ecosystem Assessment, which produced a framework for analysis in 2003 and a comprehensive analysis of ecosystem services globally in 2005, and more recently the publications of the TEEB (The Economics of Ecosystems and Biodiversity) Project (e.g. TEEB, 2010).

6.1.2. Economic valuation of ecosystem services

In the cost-benefit analytical framework, valuation is normally about comparing two different investment or policy options with different environmental (and financial) implications. In other words, valuation is about understanding the societal benefits of the difference between two (or more) options, for instance two policy options involving different levels of air pollution, rather than about understanding the value of the overall stock of environmental or ecological capital.

Economic valuation methods for ecosystem services

Two types of approaches have been developed to obtain information about the value of public ecosystem services:

- **Revealed preference approach:** uses a *link with a marketed good or service* to indicate the willingness-to-pay for the service. Examples of this type of method are the travel cost method that can be used to value the recreation service provided by an area, and hedonic pricing that can be used to value environmental attributes of goods or properties sold on a market, such as a clean living environment

- **Stated preference approach:** involve soliciting people's willingness to pay for an ecosystem service or a specific environmental quality using questionnaires or choice experiments and are of particular relevance to cultural services. Non-use values, for instance those values that people may attribute to the conservation of biodiversity without any other purpose than preserving (habitat for) specific species can only be analysed with stated preference methods, however there are significant concerns regarding their validity and reliability as discussed below.

Provisioning services	Regulating services	Cultural services
Provisioning services are private goods and changes in the supply of these services can generally be **valued on the basis of observable market prices**.	The biophysical quantification may require detailed modelling, often spatially explicit (i.e. in a Geographical Information System), of the various relevant ecological and biochemical processes in an ecosystem. These services often have a public goods character and generally require non-market valuation approaches	Both stated preference approach and revealed preference approach are used, depending on the specific service.

6.1.3. Ecosystem dynamics

Ecosystem dynamics may involve irreversible and/or non-linear and/or delayed or random changes in the ecosystem as a response to ecological or human drivers. Often, ecosystem responses are subject to thresholds, as a function of feedback mechanisms intrinsic to the ecosystem. Thresholds lead to sudden and sometimes unexpected changes in ecosystems following relatively minor increases in pressure on an ecosystem. These dynamics are critically important for ecosystem approaches, also to air pollution management, because they determine the response of an ecosystem to either an increase or a reduction in pollution loads. In addition, the supply of ecosystem services is often directly linked to the state of the ecosystem. Complex dynamics are also inherent to the RAINS (Regional Air Pollution Information and Simulation) model used for modelling critical loads in relation to air pollution. RAINS comprises a number of soil chemistry-related thresholds as well as complex responses of vegetation to changing soil conditions. Exceeding thresholds may trigger a significant ecosystem response including a change in the vitality of the ecosystem and its supply of ecosystem services.

6.2. Ecosystem Approaches and Air Quality Policy

There is a need to better understand ecosystem effects before they are included in CBAs of European air policy options. Positive effects of air pollution on ecosystem services need to be accounted for, for instance nitrogen deposition will, in N limited ecosystems, generally lead to enhanced timber production and carbon sequestration.

Given that the uncertainties involved are very significant, as outlined in this chapter, there is a particular need to conduct thorough uncertainty and sensitivity analyses to indicate the robustness of the assessments before they are used in support of policy making.

Major reductions in acidification have been achieved in Europe, in particular as result of the reduction of sulphur dioxide emissions. Based on updated data on critical loads for acidification and eutrophication for Europe, it was estimated that critical loads for acidification will be exceeded at 11 percent of the European ecosystem area in 2020, compared to 34 percent in 1990 and 20 percent in 2000 (CIAM, 2007). Nitrogen deposition, from a wide variety of sources including agriculture, will however still exceed critical loads for eutrophication in 53 percent of the ecosystem area (CIAM, 2007). There is, to date, still considerable uncertainty on how these emissions and subsequent changes in ecosystems changes have affected ecosystem services supply in Europe. The aspect that has received most attention is modelling ozone damages, in particular to crops. CIAM (2007) refers to the UK-based International Cooperative Programme on Effects of Air Pollution on Natural Vegetation and Crops (ICP), that is able to detect ozone damages across 17 European countries, and which has revealed damages of ozone in, in particular, South Germany and the Mediterranean.

6.2.1. The Gothenburg Protocol

The 2007 review of the Gothenburg Protocol carried out by the Centre for Integrated Assessment Modelling, which is hosted by IIASA, (CIAM, 2007) indicated that the Protocol bases emission reduction targets on the principle of critical loads and thresholds rather than on an ecosystem approach where the benefits of reducing pollution (due to an enhanced supply of ecosystem services) are compared with the costs (in terms of pollution control measures). Nevertheless, several preliminary figures are mentioned in the review, including damage costs for ozone. No costs for nitrogen deposition were specified. The updated CIAM (2011) study analyses the cost-effectiveness of various emission reductions scenarios to improve air quality in Europe in 2020 but does not present a further specification of the economic benefits resulting from reduced pollution in ecosystems.

6.2.2. The Thematic Strategy on Air Pollution

The Commission is pursuing work aimed at quantifying benefits from reduced exposure of ecosystems to air pollution. In particular, the Commission concluded in 2007, with Consultancy Services from ARCADIS Ecolas, a road map for enabling the monetary assessment of ecosystem benefits of air pollution abatement policies (De Smet et al., 2007). This study reviewed studies aimed at valuing benefits of ecosystem responses to reduced air pollution, and found a number of constraints. First, the number of ecosystem valuation studies is limited, and there is incomplete coverage of ecosystems and services. In particular there is a lack of studies that link (reductions in) air pollution to ecosystem services and economic benefits. Second, many dose response relations are still uncertain, and third, there is lack of information on non-use values attributed to ecosystem services. Since 2006, there is an increasing amount of literature on the valuation of ecosystem services, also in a European context (TEEB, 2010). However, there are still relatively few studies that explicitly link air pollution to ecosystem benefits (e.g. Bytnerowicz et al., 2007), and new scientific efforts to elucidate this relation can be expected, among others in the context of the upcoming Horizon2020 EU research program.

6.3. Methodological gaps and uncertainties

6.3.1. Uncertainties in ecological modelling of dose-response functions

The effect of complex ecosystem dynamics need to be better understood prior to including ecosystem impacts on CBA models. In particular, there are likely to be complexities resulting from interactions between different stressors including air pollution on ecosystems. In addition, lag effects may occur, for example reducing air pollution deposition rates to below critical load levels may not immediately lead to restoration of ecosystem functioning or ecosystem services supply. These lag effects will affect the cost benefit ratio of different policy options.

Modelling the effects of air pollution on ecosystems requires modelling of the causal chain 'emissions of air pollutant -> concentrations of air pollutants -> exposure of ecosystems -> impacts on ecosystems -> changes in ecosystem services supply'. From emissions up to the step impacts on the ecosystem, the GAINS model has been applied in support of policy making. There are as yet no European models to link ecosystem changes to changes in the supply of different ecosystem services (although models for specific services exist, e.g. for impacts on timber or crop production), see e.g. Smart et al. (2011).

The science behind the responses of ecosystems to acidifying and eutrophying components is relatively well established, and there is increasing experience with modelling the impacts of ozone on vegetation. For instance, nitrogen loading and ozone exposure cause changes in plant chemistry, photosynthesis, and ecosystem carbon balance in sensitive ecosystems. As transport and deposition of emissions continues, high N loading and air pollution (especially ozone exposure) may produce similar changes in less sensitive systems. Additional responses at these and larger scales may include shifts in dominant plant species, export of nitrates and acidity to streams, rivers, and estuaries, coastal eutrophication and harmful algal blooms and, possibly, increased invasiveness by N-demanding species (Grimm et al., 2008).

Nevertheless, there remain a number of complexities in relation to ecological dose response relationships.

Ecological dose response relationships to air pollution - Remaining complexities

1 - Air pollution affecting **different organizational levels** of biological systems including individuals, communities, species, and the ecosystem. These changes are interrelated, and differ per ecosystem, as does the ecosystem's adaptive capacity to pollution loads

2 - Effect of **multiple pollutants**: where different pollutants may enhance or reduce one another's impacts, for instance by changing the resilience to another stressor

3 - Lack of information on some of the chemical, plant physiological and plant community thresholds, which are critical for understanding ecosystem change. These **thresholds**, and the lack of detailed information on their occurrence and effects, limits the validity of dose-response relations, in particular when they are extrapolated to the European scale.

4 - Pollutant-environment interactions are complicated by the fact that **biotic and abiotic factors in ecosystems change significantly over time due to natural variations** and ecological processes. Besides oscillations on a daily basis and seasonal changes, there are long-range successional developments over time periods of decades. These variations obscure the effects of pollutants and other stressors.

Furthermore, once specific changes in ecosystem state (changes in ecosystem net primary production, species composition, etc.) as a function of changes in air pollution levels are understood, these changes need to be linked to changes in the supply of ecosystem services. This step is a lot more complex for semi-natural and natural ecosystems compared to croplands, because croplands consist mostly of monocultures with a simpler vegetation composition and structure. For other ecosystems, the scientific work on air pollution impacts has focussed on forests and water bodies (e.g. CIAM, 2007). However, the relation between pollution – ecosystem change – and supply of services has not been fully established. Changes in ecosystem service supply depend on different stressors (air pollution, water pollution, resource harvest rates, etc.), are variable due to variations in ecological processes, and strongly depend on ecosystem management. Hence, it is often very difficult to single out the impact of one specific pollutant and link this to ecosystem services supply. These complexities lead to significant uncertainties that should be considered in terms of their potential implications for policy making.

6.3.2. Uncertainties in the economic analysis of ecosystem changes

There is a need to examine how marginal costs and benefits of changes in ecosystem services supply resulting from changes in air pollution can be analysed. An important question is what effect passing critical loads thresholds will have on ecosystem functioning and subsequently the supply of ecosystem services. This effect is likely to differ for different ecosystem types and different types of ecosystem services.

There is a need to better understand society's willingness to pay for biodiversity. Reducing eutrophication, in particular, may lead to lower timber production and lower carbon sequestration in nitrogen limited forest ecosystems, but may enhance biodiversity in these forests. A question is how biodiversity effects and negative impacts on other services can be compared.

Much attention has been devoted in recent decades to the development of methodologies for the valuation of ecosystem services, in particular those services not traded in the market (e.g. Daily et al., 2009). Nevertheless, there remain significant uncertainties. These relate, in particular, to the valuation of regulating and cultural ecosystem services including the habitat service. Vatn (2005) describes a number of points of general concern regarding the valuation of ecosystem services, of which two are of particular relevance in the debate on valuing ecosystem impacts of air pollution:

(i) **lack of full information on ecosystem services**; and

(ii) **Value incommensurability**. In particular, there is still a lack of studies involving the valuation of changes in the supply of specific services due to environmental change. For instance, relatively few studies have comprehensively analysed the regulating and cultural services provided by European forest ecosystems.

In view of the large diversity, in terms of ecosystem type, ecosystem use, and socio-economic and cultural setting, extrapolating values between different sites will generally lead to a low accuracy. In addition, there is likely to occur a degree of value incommensurability in the case of ecosystem services provided by European ecosystems. This means that different types of values, for instance the values related to biodiversity, cultural functions of ecosystems and values derived from products harvested in an ecosystem, cannot be measured on one and the same scale – and that different stakeholders in different countries will attach different values to ecosystem services. This is likely to be most relevant for cultural services, where values are strongly dependent on the cultural backgrounds of the people that receive the service and may depend on religious, moral, ethical and aesthetical motives. In particular, ecosystem services that supply mainly or exclusively non-use benefits such as the habitat services (biodiversity conservation) are difficult to quantify in monetary terms, and the related uncertainties in any valuation exercise for such a service are substantial.

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Appendix 1: Uncertainties Under the Microscope

Uncertainties under the Microscope
IAM Sensitivity Scenario Analysis Can Provide a Powerful Policy Lens

A CONCAWE contribution to the AQPR

Introduction:

In the European arena a key tool that has been at the centre of air quality policy development over the past two decades has been IIASA's RAINS/GAINS Integrated Assessment Model. Both in the UN-ECE and EU context this has provided the all-important link between environmental/health impacts and cost-effective mitigation policies.

Although substantial progress has been made to make greater use of this powerful tool to explore the complete policy envelope, in CONCAWE's view more needs to be done. The purpose of this paper is to illustrate, via a number of examples, the 'policy benefits' of a thorough sensitivity analysis. Today, perhaps more than at any time in recent history, it is imperative to ensure, to the best of our abilities, that we do not unwisely expend precious economic resources in any policy arena. In the context of the current EU Air Quality Policy Review, making full use of the policy lens that GAINS provides will contribute to such a goal.

Summary of findings:

This paper was prepared as a contribution to the 4th meeting of the Stakeholder Expert Group on the EU Air Policy Review as the review enters its scenario/policy development phase. The paper is based on the results of extensive sensitivity analysis undertaken by CONCAWE using their in-house Integrated Assessment Model. This is largely based on the data IIASA developed to support their policy scenario analysis recently undertaken in the context of the revision of the Gothenburg Protocol.

The illustrative sensitivity analysis was targeted to support five contentions. Each is addressed in detail in the main section of the paper; here we provide a brief summary of the key findings:

Why the emission reductions expected of Euro VI/6 must be achieved: Policy scenarios leading to revised TSAP targets must account for uncertainties in the reductions in road transport NOx emissions associated with the introduction of Euro VI/6 standards in 2015/16. Should real-world vehicle performance result in higher than expected NOx emissions, the sensitivity analysis indicates that, at a given ambition level, this would result in significant increases in costs to the non-transport sector or even in unachievable targets. A realistic sensitivity example based on the gap closure concept as used in the CAFE 2005 program for $PM_{2.5}$ impacts, shows a factor of 3 cost increase is possible, from 7 to 20 b€/year.

Why Cost-Effective Reductions in Ammonia Emissions from Agriculture are important: Ammonia is a key pollutant; if emissions of ammonia are not reduced the scope for compensation by controls on NOx is extremely limited. It is not possible to meet ambitious acidification, eutrophication or human PM exposure targets if ammonia emissions are not reduced.

Why Multiple Time Horizons are Vital in Policy Scenarios: Policy horizon years are critical. The structural changes (e.g. changing energy use) and the on-going emission reductions resulting from already agreed legislation, have significant effects on emissions with time. This introduces the question of what is the appropriate timing for compliance with any new policy initiatives in a changing world. Investing heavily in abatement technology for the industry to achieve emissions reductions that will be reached by other means just a few years later could lead to unnecessary additional financial pressures and regret investment for industry.

Why a Range of Energy Scenario Is Important for Robust Policy: Given the uncertainties in defining the 'future world' this sensitivity analysis highlights the need for policy to be tested for a range of energy scenarios. This is vital to ensure that ambition levels (expressed as revised national emission ceilings) based on one energy scenario do not result in significant escalation in compliance costs or non-achievability in a different actual future energy world. The current difficulties in some Member States in meeting 2010 NOx ceilings illustrates the vital need to include such energy uncertainties in policy development.

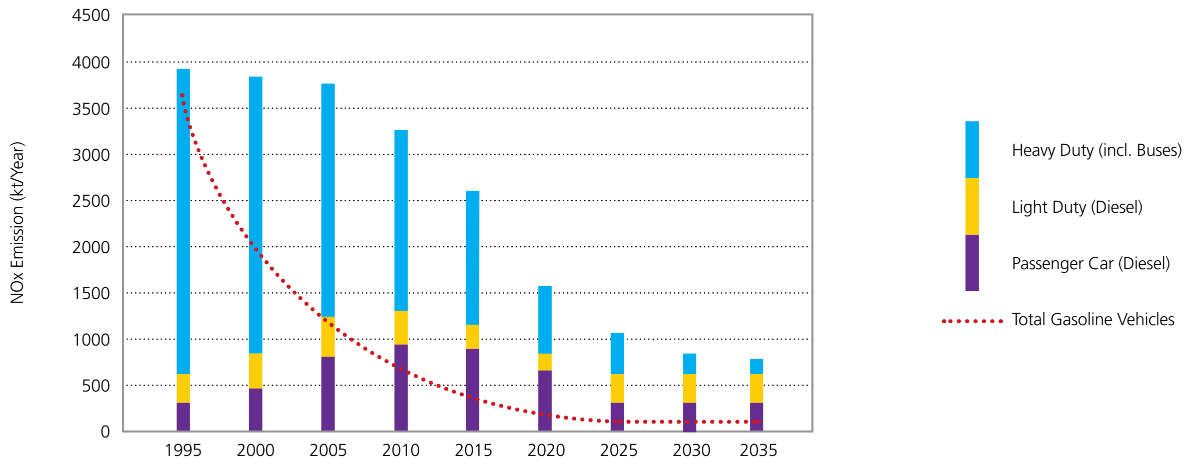
Why the influence of short Lived Climate Forcers should be more fully examined: Climate impacts of air policy need to be properly accounted for. In the context of the revision of the Gothenburg Protocol the influence of short lived climate forcers (SLCF) began to be examined in the policy process with a particular focus on Black Carbon. Other emissions such as sulphates from SO_2 and Organic Carbon are also recognized to be SLCFs. The sensitivity scenarios in this chapter demonstrate how attributing a CO_2 credit or debit to all three of these SLCF emissions (based on carbon price) and including them in the optimization strategy can give an entirely different perspective to control policies and shifts the policy emphasis away from NOx and SO_2 controls on stationary sources, even at relatively low Carbon prices and long-time horizons.

1 - Why the emission reductions expected of Euro VI/6 must be achieved:

The road transport sector remains an important contributor to overall emission levels of regulated pollutants in the EU. As such, they continue to be a priority policy target for further reductions, especially in the case of NOx. However, particularly in the case of NOx emissions derived from diesel power trains, history stands as a stark reminder of how, from Euro II/2 through to Euro V/5, real world emissions have been substantially greater than forecast from the regulated emission limits. This has led to substantial problems in achieving obligations under the current National Emission Ceiling Directive (NECD) and Ambient Air Quality Directive (AAQD) in a number of Member States.

In the context of the current Air Quality Policy Review (AQPR) process this has resulted in strong calls for Policy Makers to ensure that the planning around the Euro VI (HDV)/6 (LDV) standards is robust enough to ensure legislated limits can be met under real world driving conditions.

Figure 1
Evolution of NOx Emissions from Road Transport in EU-27: Base Case (COPERT 4)
Source: CONCAWE STEERS Model



The importance of this is illustrated by Figure 1 which shows the forecasted evolution in NOx emissions from Road Transport in EU-27 from 1995 out to 2030 and beyond. This is derived from CONCAWE's in-house road transport emissions forecasting model developed for and used extensively to support the European Auto Oil programmes. The emission algorithms (e.g., COPERT 4 emission relationships) and exogenous assumptions (e.g. fleet numbers, fleet starting vintages and turnover rates) are entirely consistent with the current version of TREMOVE used to support the transport elements of GAINS. For clarity, the trend in NOx emissions from diesel powered vehicles is shown in the stacked bars while the trend in NOx emissions of all gasoline powered vehicles is shown separately as the over-plotted red line.

What is evident from this Figure is that between 1995 and 2010 NOx emissions from diesel vehicles have not fallen at anything like the rate at which gasoline vehicle NOx has fallen. This of course is in part due to growth from the dieselisation of the light duty vehicle fleet and the general increase in vehicle kilometres driven. However, an important reason for this slower than expected reduction has been the disappointing real world performance of Euro II/2 to Euro IV/4 vehicles. Between 2010 and 2015 with the 'real world' performance for Euro V/5 already reflected in COPERT 4, this trend is not significantly changed. In contrast, by 2030 LDV diesel NOx is forecast to halve and HDV NOx reduce by eightfold from the introduction of Euro 6/VI in 2015/16 when replacement of the pre 2015/16 fleet is complete. Given past experience how can we be sure Euro VI measures will deliver such significant improvements and what are the implications of under delivery?

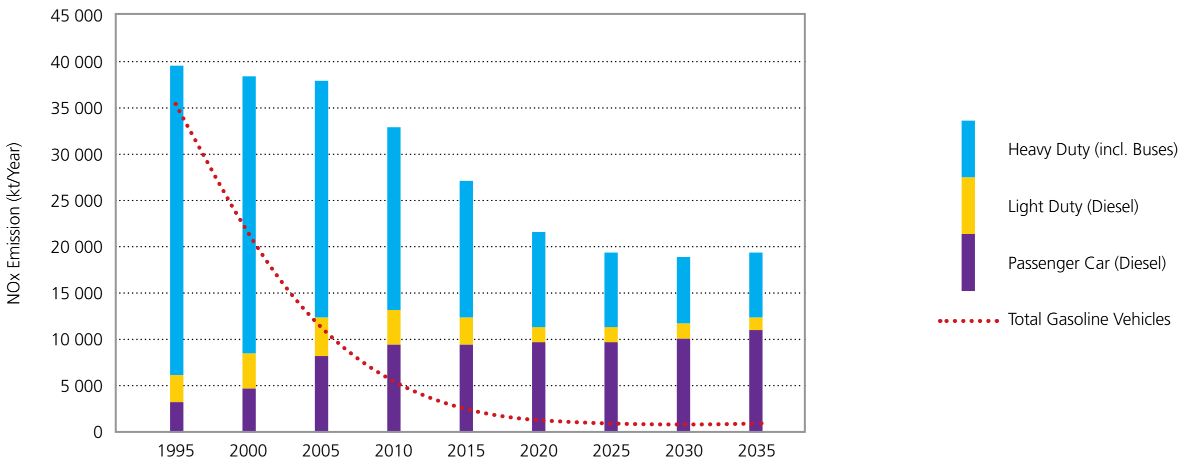
The purpose of this article is to illustrate the critical dependence of overall policy on the forecast transport NOx emissions. To undertake this we compare two emissions forecasts: one based on all vehicles achieving emissions as estimated with COPERT 4 and the other assuming higher fleet integrated emissions from the Euro VI/6 diesel fleet component. In so doing, this article does not attempt to go into any detailed considerations of how "future world" emissions from Euro 6/VI will look, especially considering the huge effort being devoted to ensuring that today's "real world" is reflected in the type approval process.

A key advantage of Euro VI/6 diesel power trains is that the standards are premised on the application of Selective Catalytic Reduction (SCR) technology which incorporates the injection of an ammonia reagent[29] to enable SCR on lean burn engines. This NOx after treatment system removes a constraint on the NOx level at the outlet of the engine, and hence allows simultaneous optimisation of engine fuel consumption through a higher thermal efficiency. The application of SCR with its NOx reduction potential (in excess of 90% for HDV and up to 75% for LDV) is thus foreseen to facilitate the simultaneous delivery of higher fuel efficiency with very low exhaust NOx. Coupled with the use of particulate filters this will also reduce dramatically primary PM from road transport.

Design of sensitivity scenarios: If sensitivity scenarios are to provide insights into the influence of uncertainties on the robustness of policies they of course must have a clear basis for their design. With this in mind the following sensitivity scenarios were constructed:

- **For Euro VI:** We have taken a sensitivity case where the fleet averaged Euro VI real world NOx emission/km would be half the emissions achievable using the Euro V emission factors in COPERT
- **For Euro 6:** We have taken a sensitivity case where the fleet averaged Euro 6 real world NOx emissions would be at the same level as the Euro 5 emissions represented in COPERT.

Figure 2
Evolution of NOx Emissions from Road Transport in EU-27: Sensitivity Case.
Source: CONCAWE STEERS Model



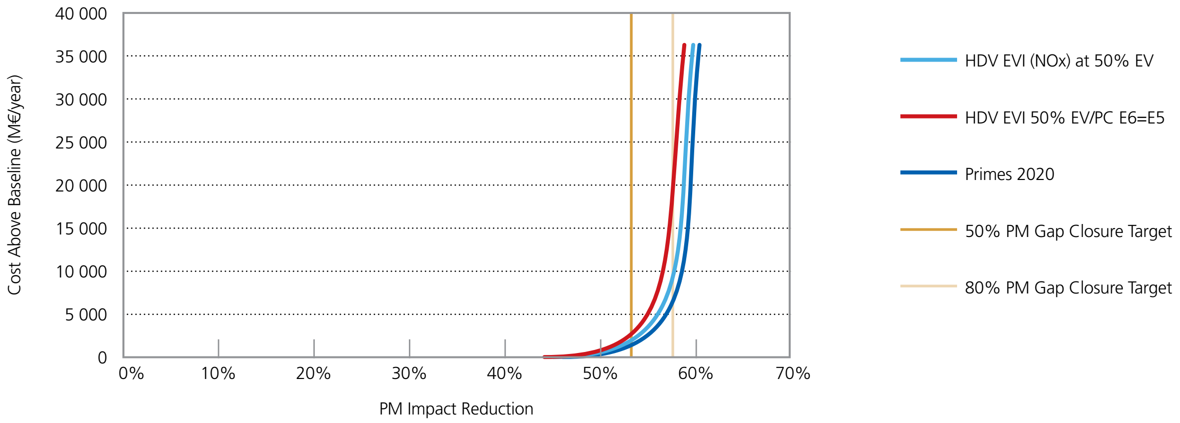
Policy Implications (e.g. Revised NECD) for higher than expected emissions from Euro VI/6 vehicles:
Figure 2 above shows the implications of the Euro VI/6 sensitivity scenario discussed above on the evolution of NOx emissions from road transport in the EU. In the 2025-2030 world of 'full penetration' of Euro VI/6, NOx emissions double over the base case, i.e. increase by some 1Mt/y.

What does this imply for NOx ceilings that are set based on the assumption that the Euro VI/6 measure does deliver forecast emission reductions?

To illustrate the policy implications, multiple optimisation scenarios were carried out using CONCAWE's in-house Integrated Assessment Model (IAM) which utilises identical source-receptor functions, cost functions and impact algorithms to those used in GAINS to support IIASA's recent work for the revision of the Gothenburg Protocol. The 'optimisation driver' was confined to PM health impacts to simplify the analysis and aid transparency. Transport emissions lie outside the optimisation as they are determined by the forecast fleet development, mileage driven and technical abatement measures in place. i.e. they are input data. The resulting optimised costs are for the additional stationary source abatement measures needed to achieve further PM impact reductions. The results are shown in Figure 3 below Note that PM impact is related to the concentrations of total $PM_{2.5}$ in the air and this comprises both directly emitted particles and secondary particles ($PM_{2.5}$ formed in the air by chemical reaction). NOx and NH_3 which we examine in the ammonia study below, contribute to secondary $PM_{2.5}$.

[29] Currently foreseen to be Urea

Figure 3
Annual Abatement EU-27 Costs For Stationary Sources (Including Agriculture) Above the 2020 Baseline Versus PM Impact Reduction
Source: Concawe IAM



Three baseline starting points were examined, all derived from the PRIMES 2009 energy scenario used as the central scenario for the revision of the GP. For the 'Base Case' the actual baseline PRIMES 2009 was used. This is shown as the dark blue line on Figure 3 and is consistent with optimised delivery of a given EU-27 PM reduction target in 2020 assuming the Euro VI/6 emissions calculated with COPERT 4. The light blue line shows the results recalculated assuming Euro VI only delivers a 50% improvement over Euro V. In this case the baseline NOx emissions were adjusted in each Member State (MS) to account for the greater transport NOx emissions before the optimisation scenarios were run. Finally, the red line shows the results assuming a future Euro VI delivers a 50% improvement over Euro V and Euro 6 is the same as Euro 5. Again, for this case, baseline NOx emissions were adjusted in each MS to account for the 'under-delivery' of Euro VI/6 before the optimisation scenarios were run.

During the Clean Air for Europe Programme, the concept of further "impact gap closure" was adopted as an indicator of policy ambition level. The '100% impact Gap Closure' being defined as the additional reduction in impacts (beyond the baseline) by implementing Maximum Technically Feasible Measures. Thus a zero gap closure is equivalent to the Baseline and a 100% gap closure is equivalent to MTFR.

The vertical lines on Figure 3 indicate the 50% and 80% PM Impacts Gap Closure points. At 50% GC, the implications for further investments in stationary sources (including ammonia abatement measures in agriculture) to make up for the greater than expected NOx emissions from road transport, should Euro VI/6 under-deliver, are already clearly significant. For the worst case considered in the sensitivity scenarios, Figure 3 shows annual costs doubling from some 1.5 b€/y to 3b€/y.

At the higher PM GC target of 80%, costs escalate since here policy would be hitting the steep part of the cost curve. In this case annual costs rise from some 7b€/y to almost 20b€/y. It is also important to note here that at the higher ambition targets, in some Member States, the resulting NOx ceilings based on the assumption that Euro VI/6 will deliver, may become unachievable even at MTFR in case of under-delivery of Euro VI/6. Such situations have already been experienced in the case of the current NECD.

What then might be a wise way forward in a policy context? Clearly this work first serves to illustrate the importance of making every 'policy effort' to ensure the next round of Euro NOx standards deliver real world emissions consistent with these standards.

But this alone is surely not enough. It is wise to explore the reductions that would be required from other sectors to compensate for a lower than expected delivery of Euro VI/6. Particularly in a context where the economies of the EU will increasingly struggle to compete in the global market place, these potential unintended consequences should be well understood. Certainly, the implications of such uncertainties (via sensitivity scenarios around the central policy case) need to be explored throughout the policy process, but especially in the final stages including their documentation in the formal 'impact assessment'.

2 - Why Cost-Effective Reductions in Ammonia Emissions from Agriculture are Important:

The Clean Air For Europe (CAFE) programme which underpinned the current Thematic Strategy on Air Pollution clearly identified the reduction in ammonia emissions from agricultural sector as an important component of cost-effective policy designed to deliver improved air quality in Europe. Through earlier policy initiatives such as the NECD and Gothenburg Protocol, the need for agriculture to be part of the solution to Eutrophication and Acidification was already well established. What was new and important in CAFE was the understanding that reductions in ammonia emissions from agriculture were central to cost-effective reductions in human exposure to fine particulates. What follows illustrates why this remains an understanding for the current AQPR and any policy initiatives resulting from this review process.

Figure 4
Annual Abatement EU-27 Costs For Stationary Sources (Including Agriculture) Above the 2020 Baseline Versus PM Impact Reduction
Source: CONCAWE IAM

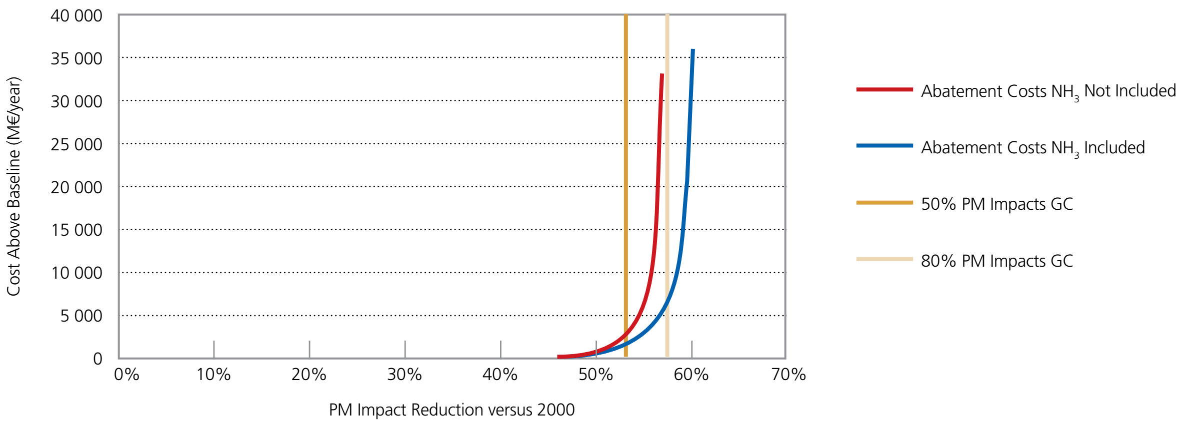


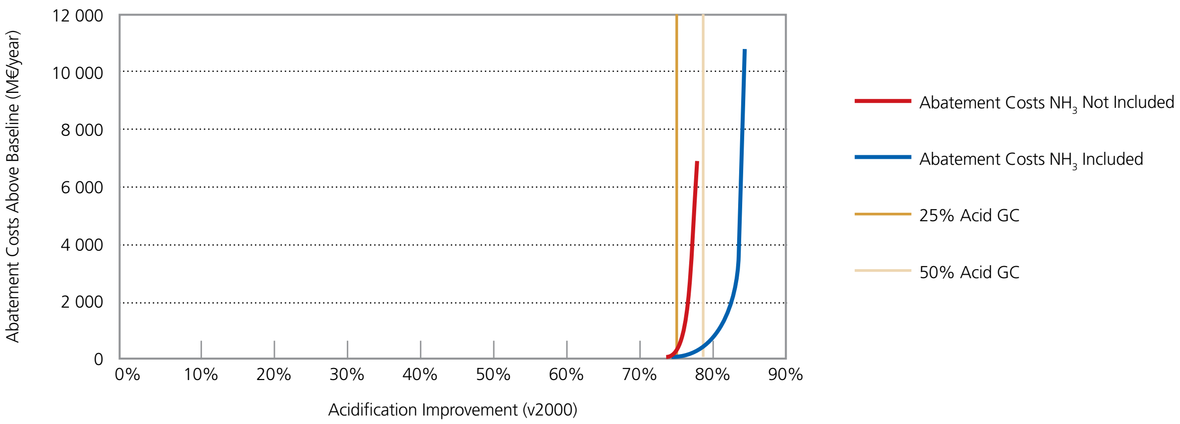
Figure 4 shows the results of integrated assessment modelling aimed at identifying the least-cost measures to deliver further improvements (beyond the Baseline) in PM health impacts in the EU in 2020. As in the work exploring the policy implications of under-delivery of Euro VI/6, this is based on the PRIMES 2009 energy scenario and associated baseline emissions that formed the central scenario for the recently completed revision of the Gothenburg Protocol.

The blue curve shows the optimised (least-cost) curve of cost versus reduction in long term health impacts of PM in the EU assuming all further abatement measures identified within the GAINS model (version used to support the GP revision work) are available for selection, including ammonia abatement measures. The red curve shows the equivalent curve but in this sensitivity case, assuming no further ammonia abatement measures are available to contribute to the cost-effective delivery of a given PM impact reduction target. In other words, ammonia emissions remain at 2020 Baseline levels.

The important, even essential contribution of reductions in ammonia in achieving optimised delivery of a given PM target is clearly evident in Figure 4. Without ammonia abatement measures, costs at the 50% PM impacts gap closure (GC) point essentially double from, some 1.5 b€/y to 3 b€/y. At the 80% GC point, this difference dramatically widens from some 7 b€/y to the MTFR point for all the 'beyond baseline' abatement for stationary sources of Primary $PM_{2.5}$, NOx and SO_2 at a cost of some 32 b€/y.

From a policy point of view, it is also worth noting that at the spend level of 7b€/y, the best achievable gap closure should ammonia emissions remain at the 2020 Baseline, is 60%. Without limit on the spend level, the best achievable gap closure, as implied above, would be 80% at MTFR.

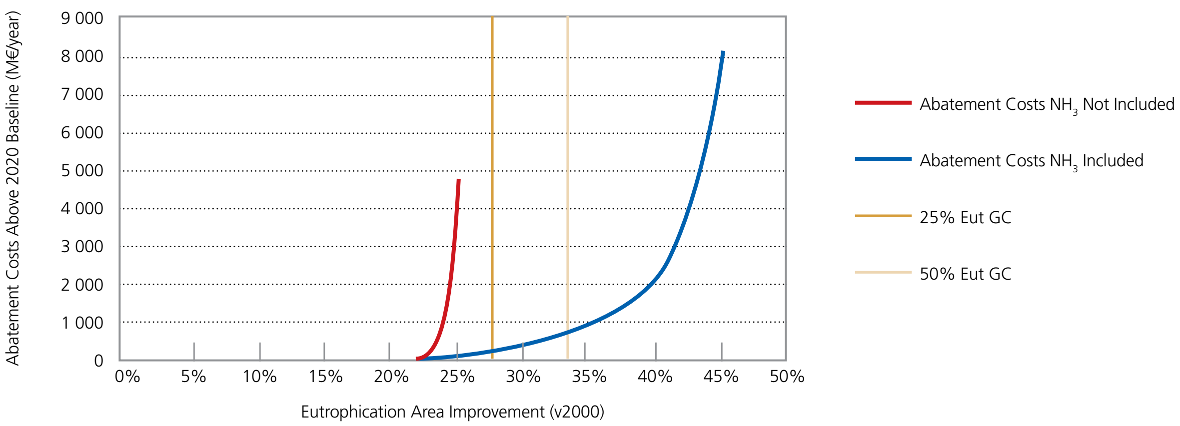
Figure 5
Annual Abatement EU-27 Costs For Stationary Sources (Including Agriculture) Above the 2020 Baseline Versus Reduction in the Area of Forest Ecosystems Exceeding Acid Critical Loads. Source: CONCAWE IAM



As already noted, ammonia reductions have long been recognised as the priority for achieving cost-effective further reductions in the areas of ecosystems exceeding acidification or eutrophication critical loads. The two curves showing the optimised cost of further abatement measure versus reduction in the ecosystem areas exceeding their critical loads in Figures 5 (acidification) and 6 (eutrophication) clearly show this.

In the case of further progress in reducing acidification (Figure 5), the maximum further improvement is severely limited if measures are confined to SO_2 and NOx. In the case of Eutrophication (Figure 6), no significant progress can be achieved without a focus on ammonia.

Figure 6
Annual Abatement EU-27 Costs For Stationary Sources (Including Agriculture) Above the 2020 Baseline Versus Reduction in the Area of Ecosystems Exceeding Eutrophication Critical Loads Source: CONCAWE IAM



To highlight the significant challenge to the policy process of ensuring the required reductions of ammonia emissions from the agricultural sector are realised, it is worth noting, in the context of the Gothenburg Protocol that ammonia emissions in the 2020 Baseline are predicted to fall by less than 2% between now and 2020. Although a new agricultural baseline scenario is under preparation, the optimisation undertaken in this 'GP PRIMES 2009' scenario, foresees the cost-effective contribution to the 50% PM GC target to result in a 17% reduction from 'today's' level and a 29% reduction in the case of an 80% PM GC target. A challenge indeed!

3 - Why Multiple Time Horizons are Vital in Policy Scenarios:

In the policy context of a revision of the TSAP with horizon years out to and possibly beyond 2030, the need to consider the on-going influence of already agreed policies (for example changes induced by structural change driven by climate policy, turnover of the vehicle fleet) is vital. This requires a focus on several policy horizon years. What follows is designed to illustrate the economic importance of such a focus and is based on recent GAINS cost curve data for 2020, 2025 and 2030.

Figure 7
National NOx Cost Curves for Three Policy Horizon Years for One Example Member State
Source: IIASA for 2012 TSAP

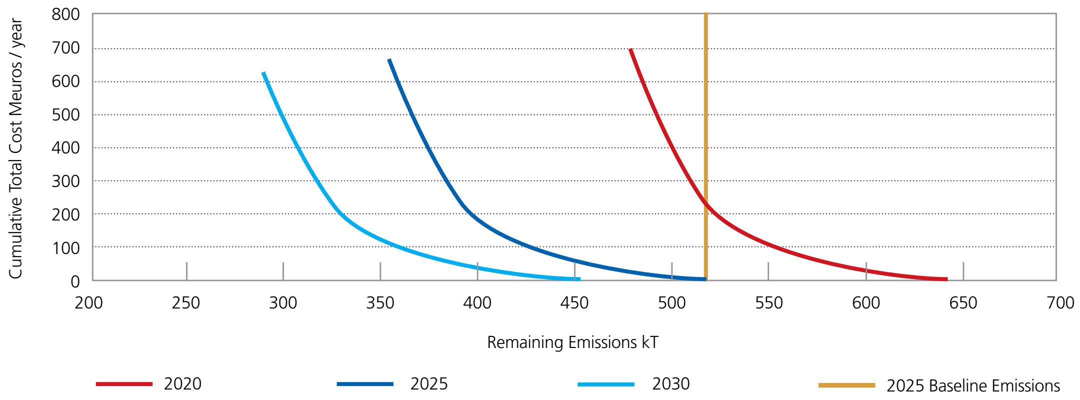
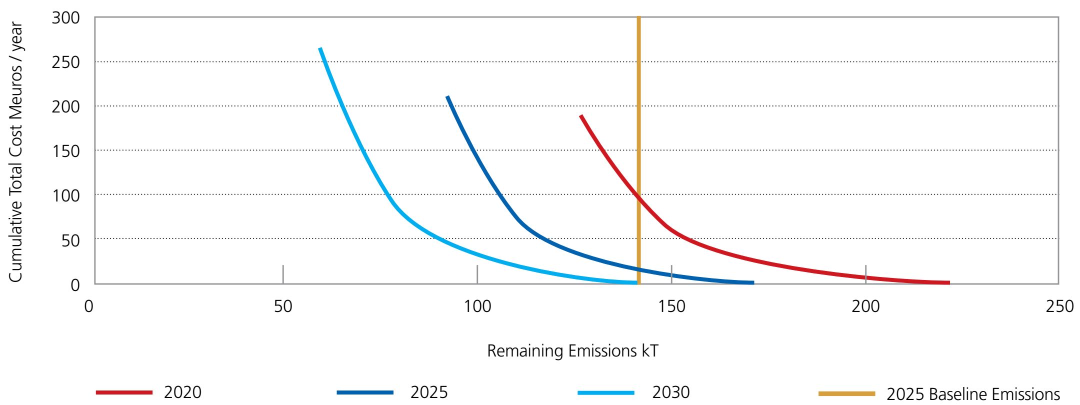


Figure 7 compare national cost curves for further abatement measures on NOx in one example Member State of the EU for three policy horizon years: 2020, 2025 and 2030. These are based on IIASA GAINS data developed for their current work on the revision of the TSAP. In each case, the continuing effects of base case changes in emissions are clearly seen. This has significant implications for the cost of achieving further impact reductions as a function of time.

If, for example, GAINS indicated that the revised TSAP targets required this Member State to reduce its baseline NOx emission to 515kt/y this MS would be faced with an additional cost burden for NOx reducing measures of some 250 M€/y if the targets were required to be met in 2020. However, should the time horizon for achieving the target be moved out by five years to 2025, baseline measures would deliver the ceiling without further burden to that MS.

This continuing influence of already agreed policies on emissions versus time is not confined to NOx as indicated by the corresponding cost curves for SO_2 in the same example MS given in Figure 8.

Figure 8
National SO_2 Cost Curves for Three Policy Horizon Years for One Example Member State
Source: IIASA for 2012 TSAP



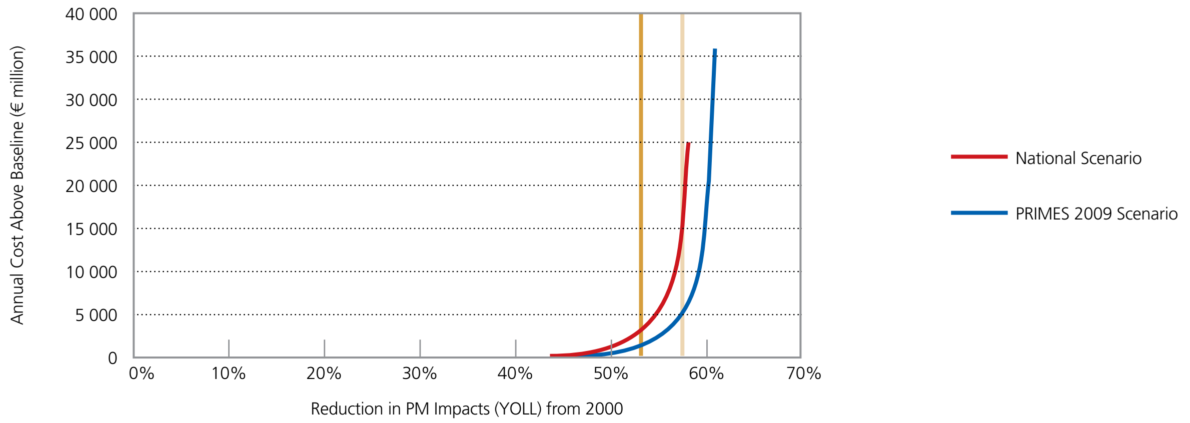
Of course in looking at future policies designed to make further progress in air quality in the EU it is also important to recognise the on-going costs of already agreed measures which are delivering these continued reduction in baseline emissions (with their associated further improvements in air quality) with time. For this example MS for NOx alone, GAINS indicates the cost of already mandated measures in 2010 to be some 2.8 b€/y, rising to 5.3 b€/y in 2020 and reaching 6.7 b€/y by 2030.

4 - Why a Range of Energy Scenarios Is Important for Robust Policy:

The need for consistency/coherency in the central assumptions used in the development of interrelated policy initiatives (e.g. Air Quality and Climate Change) is well recognised. However, this should not be interpreted as a need to base policy on a single view of the 'future world' that the policy is designed to influence. History serves as a constant reminder that actual developments can be quite different from the projections made a few years earlier. Sensitivity scenarios around a central view to test the robustness of future business plans are essential to the business world. In CONCAWE's view such sensitivity analysis is also essential in the policy arena.

In this regard, along with a number of other stakeholders, CONCAWE has requested that a range of energy scenarios, around the central PRIMES scenario, should be used in appropriate sensitivity scenarios to test policy options. In this short section, the databases used for the revision of the Gothenburg Protocol have been used to support this call.

Figure 9
Annual Abatement EU-27 Costs For Stationary Sources (Including Agriculture)
Above the 2020 Baseline Versus PM Impact Reduction: Comparison of PRIMES and 'National' Energy Scenarios Source: Concawe IAM



Although only twelve Member States submitted their alternative national energy scenarios, the consequence of moving from a PRIMES based world to this alternative 'National Energy Scenario' world is already significant. Figure 9, shows the optimised curves of cost beyond the baseline versus further reductions in PM impacts for each energy scenario. The two vertical lines indicate a medium (yellow) and high (red) improvement target. The implications of arriving in the 'National energy future world' having designed policy with a sole focus on the PRIMES world are obvious: costs, justified only for the PRIMES world, double at the medium ambition level and triple to close to MTFR costs at the high ambition. In the latter case, at an individual Member State level some individual pollutant ceilings set solely based on PRIMES would likely at this ambition be unachievable. Given the binding nature of the NECD, this would force Member States to consider measures that would otherwise not be justifiable and could have undesirable economic consequences. Such a situation would be avoided with the inclusion of suitable sensitivity analysis at the policy development phase.

5 - Why the influence of short Lived Climate Forcers should be more fully examined:

One key recent development in the context of the revision of the Gothenburg Protocol was the inclusion of considerations over the influence of short lived climate forcers (SLCF) in the policy process with a particular focus on Black Carbon. As a consequence, the GAINS team have begun to incorporate such considerations in a quantitative way into GAINS.

What this work by IIASA has provided is a helpful bringing together of quantified data on the direct greenhouse warming potential (GWP) of all the key SLCFs and was first presented by IIASA in Dublin in May 2010[30]. The following data for GWPs have been abstracted from this presentation:

Table 1
Global Warming Potentials relative to CO_2 (GWP CO_2=1)

	20 year GWP	100 year GWP
SO_2	-140	-40
Black Carbon	2200	680
Organic Carbon	-240	-75

The availability of these relative GWPs allow the "CO_2 compensation costs" implied for a unit reduction in each of the three SLCF to be computed for a given carbon price. The carbon compensation cost here is the cost involved in sustaining 'no change' in Baseline GWP by introducing compensating measures. In the case of SO_2, since this is a climate cooler, at a carbon price of 30€/tCO_2, this would imply carbon compensation costs of 4200 € for every tonne of SO_2 emissions reduced (assuming a 20 year integration period) and 1200 €/tSO_2 over a 100 year integration period. In other words, removing the beneficial climate cooling effect of sulphates derived from SO_2 emissions has to be compensated by additional climate mitigation measures. Conversely, in the case of black carbon (a powerful warmer), for the same carbon price the compensation cost would be -66,000 €/tBC and -20,400€/tBC over the two integration periods. In other words, reductions in emissions of this powerful climate warmer result in savings in the climate mitigation costs of the baseline.

The availability of these CO_2 compensation costs provides a means of more fully expressing the implications of air quality policies that results in further reductions in these pollutants. For example in the case of measures PM abatement measures, the reduction in CO_2 abatement costs implied by attendant reduction in the black carbon fraction of PM can be quantified. Similarly, for SO_2, the implied additional CO_2 compensation cost for removing this 'climate cooler' can be quantified.

By building these 'CO_2 compensation' costs in the form of adjustment algorithms to the basic cost curves derived from GAINS, these costs can then be accounted for in the optimisation strategy to derive a more complete 'least cost' set of measures that delivers the air quality objective accounting for the CO_2 compensation costs. Based on detailed data kindly made available by the GAINS team in the context of the GP revision process, CONCAWE have recently built this capability into their in-house IAM. What follows are some first results which indicate the importance of taking the full implications of SLCF into account in developing future policy. Importantly, the work clearly indicates that the inclusion of the considerations into the optimisation strategy significantly shifts the policy emphasis away from further controls for SO_2 and NOx on stationary sources, even at relatively low carbon prices and long-time horizons.

Figure 10 shows the additional cost of stationary source measures (beyond 2020 baseline) for a number of PM impact reduction targets. The costs are shown for each pollutant. Here the optimisation strategy did not include the CO_2 compensation cost for SO_2 and the Organic Carbon (OC) content of $PM_{2.5}$ emissions. Nor did it include the savings in CO_2 mitigation cost in the baseline derived from any reductions in Black Carbon (BC) emissions. Figure 11 shows a repeat of the same analysis, but in this case the SLCF compensation costs were included in the optimisation strategy[31]. In both figures the net costs (abatement costs plus CO_2 compensation costs) are shown as the grey area. As in earlier sections, this analysis has been based on the PRIMES 2009 GAINS data set used for the central policy analysis for the recent revision of the Gothenburg Protocol.

[30] First presented by Markus Amann at the 38th session of the UN-ECE TFIAM meeting in Dublin, May 17-19, 2010
[31] With a 2050 target date for Climate Stabilisation in view and a 2020 policy horizon for delivering the PM impact reduction target, an integration period of 30 years was used for the relative GWPs of SLCFs compared to CO2. These were determined by linear interpolation of the data in Table 1.

Figure 10
Annual Abatement Costs for EU-27 by Pollutant For Stationary Sources (Including Agriculture) Above the 2020 Baseline versus PM Impact Reduction Including Carbon Compensation Cost for SLCFs Case 1: 30 €/tCO₂ and SLCFs Compensation Costs not in Optimisation Source: Concawe IAM

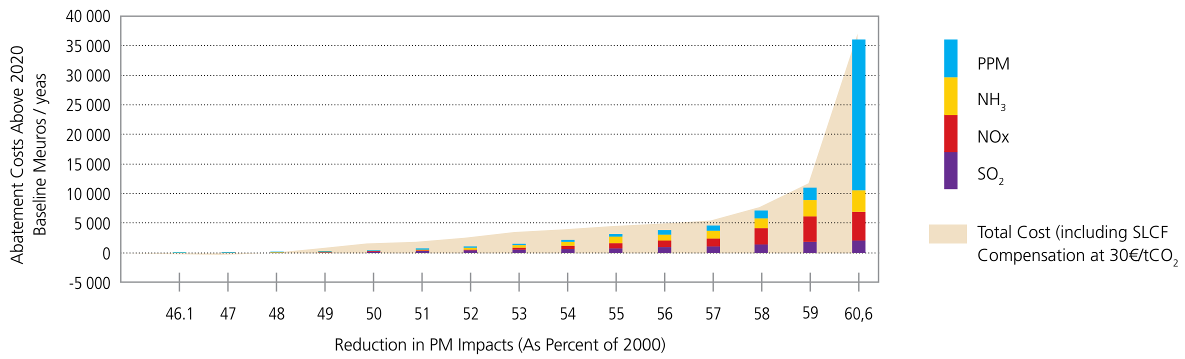
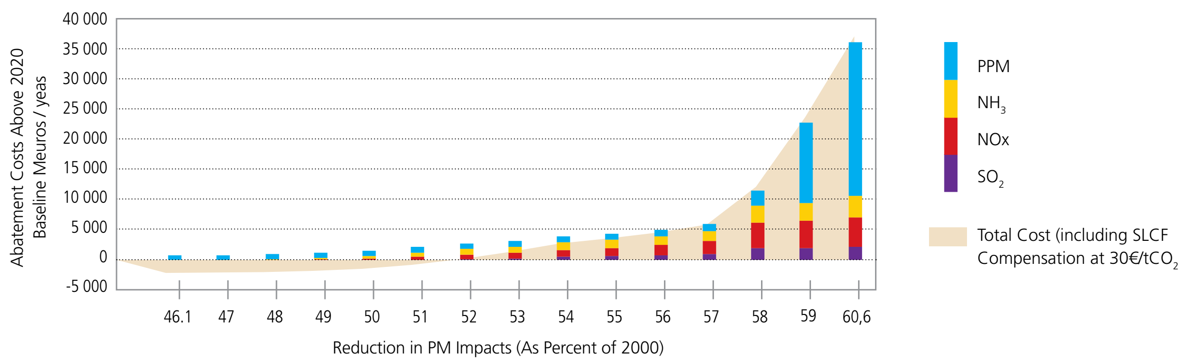


Figure 10 shows how the optimiser, at least up to the 54% improvement target, picks abatement measures on SO_2, NH_3 and NOx rather than primary $PM_{2.5}$ reducing measures. As may be seen, this results in a significant additional cost of measures to compensate for reductions in SLCF (here mainly SO_2).

At the 54% point, the cost of further abatement measures would be some 1.5b€/y. To this needs to be added the implied CO_2 compensation cost, which, assuming a carbon price 30€/t CO_2, would be some 2.1b€/y. Thus the net cost, as shown on Figure 10, would be 3.6b€/y. At and beyond the more ambitious improvement target of 58%, most SO_2 measures have been exhausted and the optimiser picks Primary $PM_{2.5}$ measures. Since these emissions include a black carbon component, their reduction results in savings in the cost of climate mitigation measures included in the baseline, and the difference between abatement cost and net overall costs reduces.

An important policy perspective emerges when SLCF compensation costs are included in the optimisation. The results are shown in Figure 11. What is immediately clear in Figure 11 (compared to Figure 10) is that optimiser first targets primary $PM_{2.5}$ abatement measures with a high fraction of BC component. This is not surprising based on the relative GWP for BC given in Table 1. Using the 20 year integration period value of 2200, a carbon price of 30€/tCO$_2$ and a BC content of $PM_{2.5}$ of 50% yields a compensation cost of -33,000/tPM$_{2.5}$. If the cost of $PM_{2.5}$ abatement in such a case was 15,000€/tPM$_{2.5}$ the net cost for the measure would be a cost saving of 18,000 €/tPM$_{2.5}$ and that measure would be picked by the optimiser as a 'first pick'. This is why, at the lower end of the improvement target range in Figure 11, net costs are negative.

Figure 11
Annual Abatement Costs for EU-27 by Pollutant For Stationary Sources (Including Agriculture) Above the 2020 Baseline versus PM Impact Reduction Including Carbon Compensation Cost for SLCFs Case 2: 30 €/tCO₂ and SLCFs With Compensation Costs in Optimisation Strategy. Source: Concawe IAM



Importantly, while net costs remain negative up to the 53% improvement target, abatement costs themselves are clearly higher than those shown in Figure 10. In other words, as well as moving away from measures controlling secondary sources of $PM_{2.5}$, the overall abatement burden on some sectors would increase.

Finally, the shift to focussing on black carbon rich PM abatement measures is consistent with the emerging evidence, at least from toxicological studies, that the black carbon fraction of PM is likely to be a more potent actor than the secondary component in impacting human health.

In CONCAWE's view, these first results serve to demonstrate the importance of accounting for SLCF in the context of the current Air Policy review process.

Appendix 2: CBA Under the Microscope

Cost Benefit Analysis under the Microscope

CONCAWE Comments on the Key Submissions Associated with 5th Stakeholder Expert Group of the Air Quality Policy Review held in Brussels, 3rd April 2013

General points

Cost benefit analysis (CBA) seems to be increasingly referred to as a basis to support target setting for air quality policies when in the past (CAFE 2005 program) its function was to provide an ex-post perspective on costs and potential benefits associated with the delivery of the policy ambition levels. With CBA the societal costs and benefits of different ambition levels can be compared, provided that both costs and benefits are expressed in a monetary unit. Recent CBAs conducted in support of European air quality policies have focussed on comparing costs and benefits of 5 specific scenario's (CIAM, 2011), each comprising a mix of targets for reducing the ambient concentrations of PM, ozone, acidifying and eutrophying substances. These studies suggest that the monetised benefits of air pollution control exceed the costs of emission reduction, for all of these five scenario's. The benefits are driven by the particular value given to the statistical improvements in average life expectancy arising from reduced exposure to fine particulates.

Since CBA is having an increasing role in the target setting process of the current Air Quality Policy review, it is crucial that it is applied in a scientifically robust manner. CONCAWE sees that there are at present several important flaws and limitations in the way CBA is applied in the Air Quality Policy review process. This paper describes the main limitations and proposes a more robust approach for VOLY calculation from a given Willingness to Pay (WTP) survey. Three of the limitations are specific methodological inaccuracies in the CBAs conducted in support of the EU air quality policy formulation process. All three aspects have very significant implications for the outcomes of the CBAs of air policy targets and should be addressed as part of the current policy formulation process. Two additional limitations are fundamental to CBA. They cannot be addressed through specific methodological upgrades, but they need to be kept in mind in the interpretation of CBAs.

Important issues related to CBA that should be addressed in the current phase of air quality target setting

In CONCAWE's view, the major shortcomings in recent CBAs (AEA, 2011; EC4MACS, 2011; EEA, 2011) conducted in support of the Air Policy review process that prevent the work from providing robust policy input are: values for monetising (i) mortality and (ii) morbidity effects, and (iii) an insufficient uncertainty analysis to analyse the repercussions of these uncertainties on the costs and benefits of different policy targets. Regarding point (i) CONCAWE proposes a more robust approach to express a single value to represent the results of a given WTP survey. This is outlined below.

(i) Improved statistical life expectancy – Use of a more robust VOLY value

Estimating the monetary value of a life year in a given population is not an easy thing to do. The chosen method is to survey people for their WTP to achieve a small increase in statistical life expectancy (see CONCAWE, 2012b). It must be kept in mind that the responses represent 'virtual' rather than 'real' money, are very varied and depend on the size of the risk reduction assessed. The highly skewed distribution of responses (ranging by over three orders of magnitude and highly skewed towards the low end (Figure 1) should be used directly in Monte-Carlo analysis of cost-benefit after rescaling to represent one life year increments. However policy-makers prefer a single reference "VOLY" for ease of communication. The CBAs conducted for the Air Quality Policy review (AQPR) uses the median value from the NewExt study (2004), updated for inflation (to match costs) and without regard to whether attitude has changed in the ten years since the survey was carried out. We note that the median does not respect individual WTP choices; it just marks the dividing price for the risk reduction where 50% say they are not willing to pay more for the statistical benefit.

As stated above it was acknowledged by the CBA community during the Clean Air For Europe (CAFE) program that the most representative CBA results could be obtained by statistical analysis using the full distribution of WTP survey results and distribution of abatement costs together as discussed in detail in the CONCAWE report 4/06. The challenge in shortening this process to use a single representative value is to find a means of respecting individual choices. CONCAWE's recent work is a response to this challenge (CONCAWE, 2012a). This alternative approach defines a VOLY that Maximises Societal Revenue (MSR), while respecting individual expressions of WTP of all the individuals surveyed. This is achieved by a simple flat fee analysis to determine VOLY from a WTP survey, it assumes a pay/no pay threshold and sets the threshold (fee) to maximise the sum that can be raised from the survey population. This is a technically and methodologically more robust approach compared to using median or mean values of a survey. The biggest advantage is that this flat fee approach reflects the full distribution of expressed WTP values and is less sensitive to the very highest and lowest choices. How the MSR values relate to the median and mean values is shown in Table 1 and Figure 2 for a number of elicited WTP studies. It is worth noting that the MSR values correspond more closely to the median and are far away from the mean VOLY values of each of the surveys.

Figure 1
Highly skewed distribution of WTP values as forecasted from the Weibull distribution parameters of the NewExt survey in 2004 (CONCAWE, 2006a)

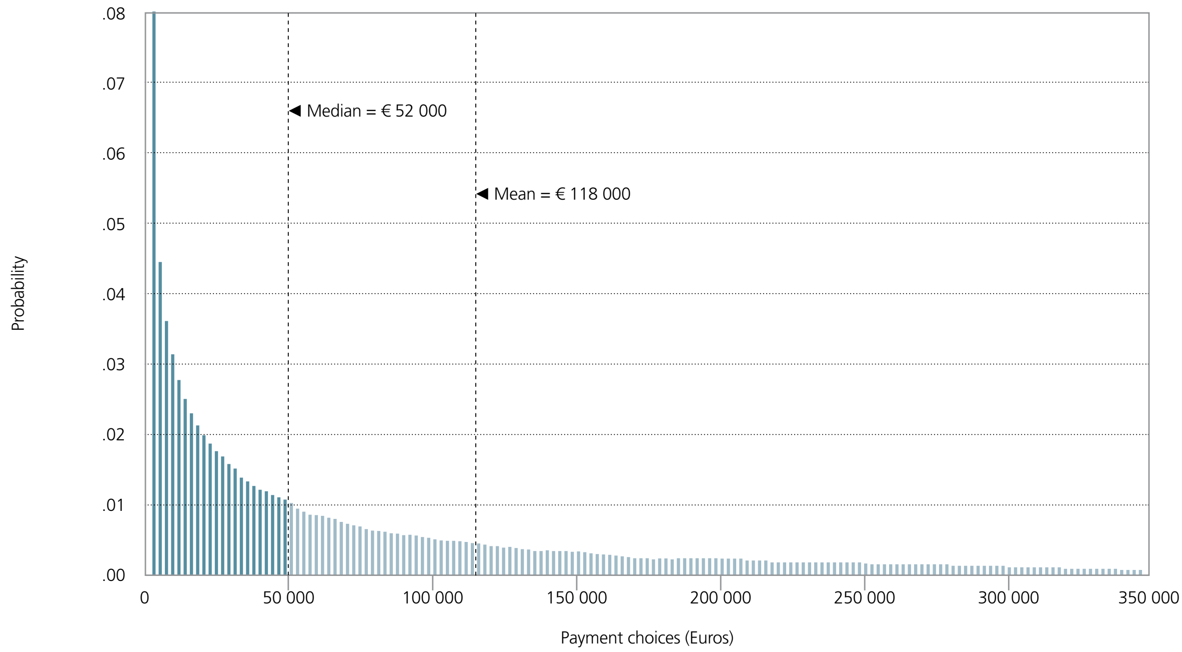


Table 1
VOLY values (€ per statistical life year lost) for a number of elicited WTP studies

Study	VOLY Median	VOLY Mean	VOLY based on MSR/4
AEA (2011)/1	57,700	138,700	37,000
EC4MACS (2011) / EEA (2011)/1	54,000	125,000	37,000
Desaigues et al. (2011) - 6 months/2	15,200	24,700	9,100
Desaigues et al. (2011) – 3 months/2	19,400	38,400	13,000
DEFRA - 6 months/3	2,700	13,000	3,400
DEFRA - 3 months/3	2,200	23,000	5,500
DEFRA - 1 month/3	15,000	45,000	13,000

Note /1: based on the NewExt (2004) study but corrected for inflation.
Note /2: based on the NEEDS study (equation 1; Desaigues et al., 2007) but corrected for inflation (population weighted average for 18 EU countries plus Switzerland).
Note /3: Note that '1 month', '3 months' and '6 months' refer to the different risk-reduction choices in these WTP studies. The values represent averages of assessments for normal health for each risk-reduction choice and are not corrected for inflation; Chilton et al., 2004.
Note /4: CONCAWE, 2012a.

Figure 2
The (forecasted) VOLY distributions (20 to 80 percentiles) according to three studies, also indicating the location of the median, mean and MSR value

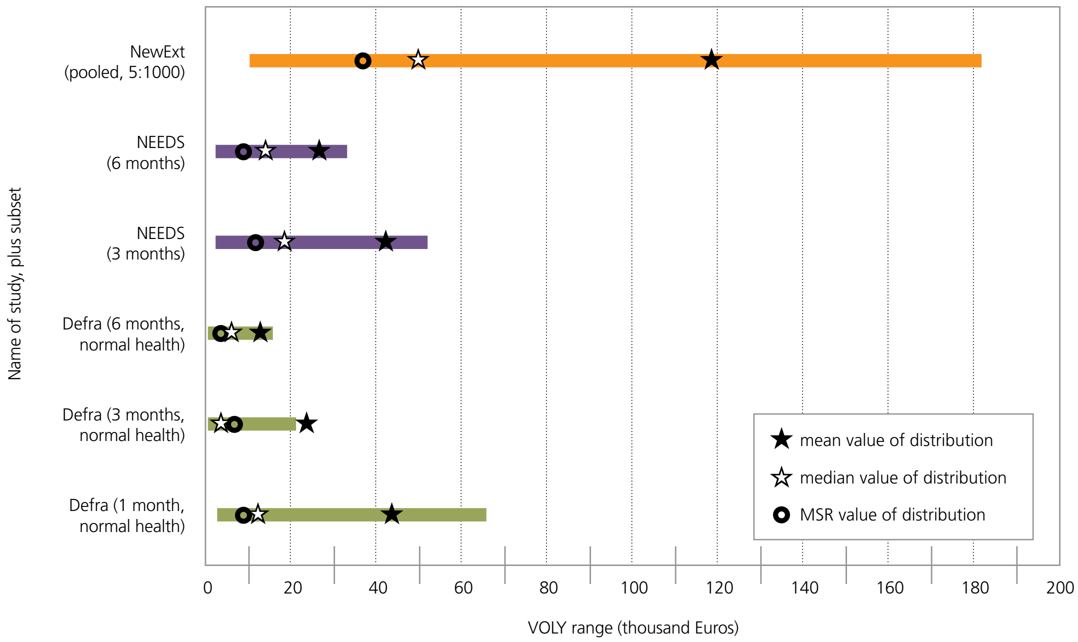


Table 1 and Figure 2 clearly show that VOLY values across a number of elicited WTP studies are very different and that they are dependent on the risk reduction discussed as shown by the variation in annualised values between the one month, three or six months increase in life expectancy (Desaigues et al. and DEFRA).

All of the analysed recent CBAs (AEA, 2011; EC4MACS, 2011; EEA, 2011) that were conducted in support of the Air Policy review process on behalf of the European Commission are based on a single VOLY estimate prepared by the NewExt project that was used during the CBA work for the Clean Air For Europe (CAFE) process. The NewExt study (2004) resulted in a median VOLY of €52,000 and a mean value of €118,000. It must be noted that the NewExt developed data for prevented fatality (VPF), i.e. analysing the Value of a Statistical Life (VOSL), to derive a VOLY value an inappropriate methodology was applied by back-calculating from VPF/VOSL, rather than estimating the value of VOLY directly based on WTP surveys. This is fully elaborated in earlier CONCAWE work (2006a & b).

Since the NewExt study was conducted a decade ago there are a number of new insights in the field. As recognised in the NEEDS study, which was a follow-up to NewExt, the VOLY should be derived directly from survey questionnaires rather than be derived from the VOSL (see also CONCAWE, 2006a & b). In addition, there is increasing experience with the design of questionnaires used to elicit WTP. These new insights were reflected in the NEEDS study, that was published in 2011 in the scientific literature (Desaigues et al., 2011). It is notable that some of the researchers involved in NewExt were also involved in the NEEDS study. The NEEDS study points to much lower median and mean VOLY values, of €15,200 and €24,700, respectively (Table 1, Figure 2; published in Desaigues et al., 2011). These recent findings are much more in line with other research studies (e.g. the DEFRA study conducted in the UK).

However, all recent CBA studies (EC4MACS, 2011; EEA, 2011; AEA, 2011; IIASA (TSAP report #7), 2012) have continued to use the inappropriate VOLY from the NewExt study, simply adjusting its values for inflation in the geographical zone of the individual study (inflation corrector differs as a function of different geographical scope of the studies because of different inflation rates for each country). This is reflected in median and mean VOLY values of €54,000 or €57,700 and €125,000 or €138,000, respectively (see Table 1). This has far reaching consequences for the benefits calculation of air pollution reduction measures as the VOLY dominates the benefits associated with CBA, i.e. reduced statistical life expectancy from exposure to PM represents around 70 to 75% of the total benefits.

CONCAWE strongly believes that the VOLY values derived from the NewExt are inappropriate and should not be considered in CBA analyses for the ongoing or any future air quality policy rounds. Instead we would propose that an (weighted equally across the NEEDS and DEFRA surveys) averaged VOLY value, based on both the NEEDS and DEFRA surveys be used. Based on the more robust MSR methodology, a value of €9,250 (Table 2) should be considered the reference value given that it represents the most up-to-date and more scientifically robust estimate for a VOLY. The range of MSR VOLY values from €3,400 to €13,000 (Table 2) should be used for a sensitivity analysis for further verification of the robustness of the CBA of the current TSAP review process.

Table 2
Appropriate VOLY expression (€ per statistical life year lost) of more suitable WTP studies (no correction for inflation applied)

WTP study	VOLY Median	VOLY Mean	VOLY based on MSR/3
NEEDS - 6 months/1	14,000	27,000	9,100
NEEDS - 3 months/1	19,000	42,000	13,000
DEFRA - 6 month/2	2,700	13,000	3,400
DEFRA - 3 months/2	2,200	23,000	5,500
DEFRA - 1 month/2	15,000	45,000	13,000
Weighted average VOLY of all studies	**11,600**	**31,000**	**9,250**

Note /1: Chilton et al., 2004.
Note /2: Desaigues et al., 2007. Note that '1 month', '3 months' and '6 months' refer to the different risk-reduction choices in these WTP studies. The values represent averages of assessments for normal health for each risk-reduction choice.
Note /3: CONCAWE, 2012a.

Recommendation: If a single value is adopted to describe such WTP surveys, then MSR is a more robust approach as it respects individual expressions of WTP of all respondents to the survey. As such it reflects the full distribution of WTP survey results and reduces the dominance of more extreme values. Disregarding the VOLY values of the NewExt study as this WTP survey was not designed to derive a VOLY value in first place, the MSR approach gives an (weighted) average VOLY value of €9,250 (not corrected for inflation), based on the NEEDS and DEFRA WTP studies. This value is considerably less than the €54,000 to €57,700 used in current policy developments. When applying a sensitivity analysis the (weighted average) range from €3,400 to €13,000 (not corrected for inflation) should be tested.

In light of the above, CONCAWE believes that there is a strong case for adopting a MSR VOLY averaged over all suitable WTP studies (i.e. €9,250). This compares to a median VOLY averaged over all suitable WTP studies of €11,600.

(ii) Realistic and sound reflection of morbidity effects

There is a need to re-assess the dose-response relations for morbidity effects and the monetary values of these effects. In all the recent CBA studies the dominating component of overall benefits has been the reduced statistical life expectancy associated with exposure to particular matter (PM; i.e. around 75%). Therefore, there has been little attention to the uncertainties associated with other elements of monetised benefits, such as morbidity effects. However, the valuation of these effects may also be overestimated (IIASA, 2012), in particular for chronic morbidity and restricted activity days (RAD)[32], as an analysis by CONCAWE indicates (see Attachment 1 "Technical Note. An Assessment of the Valuation Methods for and Costs of Morbidity" for details).

For instance, the costs of chronic bronchitis have been estimated at €208,000 per case (IIASA, 2012). This cost figure applied during the CAFE program is based on two studies: Viscusi et al. (1991) and Krupnick and Cropper (1992). This value is also cited in several recent European CBAs on morbidity impacts, while there is a set of more recent studies available in the scientific literature and despite the fact that already in the CAFE program it was noted that the two studies (Viscusi et al. 1991 and Krupnick and Cropper 1992) use a definition of chronic bronchitis that is not in line with the epidemiological literature. An assessment by CONCAWE of the more recent studies that consider both the costs of medication and the WTP to avoid the symptoms of chronic bronchitis reveals much lower values (Menn et al., 2012; Maca et al., 2011; Chapman et al., 2006; Stavem, 2002; Wilson et al., 2000; Priez and Jeanrenaud, 1999; O'Conor and Blomquist, 1997). A reasonable approximation of the value of chronic bronchitis may be in the order of € 28,000 to € 38,000 per case. This is approximately 1/6th to 1/7th the value used in the most recent European assessments of the costs of air pollution, such as IIASA (2012). It is clear that there is an urgent need for verifying the morbidity costs of the recent CBAs and that the values currently used should be interpreted with caution.

[32] According to the CAFE methodology, RAD include (i) days when a person needs to stay in bed and (ii) days when a person stays off work. Days including other, less serious, restrictions on normal activity are called minor RADs, and are valued separately

With regards to Restricted Activity Day (RAD), there are important uncertainties in relation to the assumed monetised value as acknowledged in Ready et al. (2004) but not in CAFE or the follow-up CBAs. As based on CONCAWE's preliminary assessment, these uncertainties are:

1 - The value is based upon just one study (Ready et al., 2004)
2 - The results of this one study (Ready et al., 2004) are internally inconsistent, specifically the outcomes that the WTP of Spanish and Portuguese to avoid a RAD far exceeds that of northern Europeans, and that the WTP to avoid a minor RAD exceeds that the WTP to avoid a 'regular' RAD in Spain. This may be related to an incorrect application of the survey.

While it is clear that the monetised RAD values used in CAFE are likely to be overstated, it is difficult to say how much the costs of RAD would be overestimated as data are missing. It seems highly questionable if it is justifiable to use monetary values with such a high degree of uncertainty as a basis for decision making and hence we suggest to remove this from the benefits analysis until a proper sensitivity analysis has been performed.

Recommendation: As expressed by CONCAWE's preliminary analysis it is clear that there is an urgent need for verifying the morbidity costs of the recent CBAs and that the values currently used should be interpreted with caution. In particular, values of chronic bronchitis and RADs are based on very few valuation studies, based on limited sample surveys, and with inconsistent results for some of the studies. Based on the recent studies the analysis also reveals a more up-to-date value range for monetising new cases of chronic bronchitis from €25,000 to €28,000 per case rather than the €208,000 currently applied in CBAs. Furthermore given the uncertainties around monetising RADs this end point should be removed from the benefit analysis until a proper sensitivity analysis has been performed.

(iii) Conduct proper uncertainty analysis

Furthermore, an issue of concern is that there is at present insufficient understanding of the effects of the various sources of uncertainty on the outcomes of the CBAs. These uncertainties pertain to: uncertainties in future baseline emissions as a function of economic developments, uncertainties in future energy prices and impacts on future emissions, uncertainties on ambient concentrations as a function of emission reductions and other factors (including weather patterns and wind directions that may be influenced by climate change), uncertainties in exposure levels of people as a function of human behaviour, and uncertainties on the monetary value of health and environmental effects. CONCAWE has earlier provided input to the TSAP review process with regards to this discussion (CONCAWE, 2012c).

Relatively minor changes in each of these factors could have significant repercussions for the cost benefit ratio of different emission control strategies. In order to ensure the selection of no-regret policy measures, there needs to be an understanding of how robust the cost and benefit assessments are prior to air quality policy setting.

Recommendation: In addition to using an updated VOLY figure (i.e. €9,250), there should be a proper uncertainty analysis, for instance using Monte Carlo analysis with full distributions of both benefit and cost in order to analyse the repercussions of these uncertainties on the overall costs and benefits of different policy targets.

Two fundamental aspect to keep in mind when using CBA for policy formulation

In addition to these three key issues that need immediate attention as part of the current policy formulation process, there are two aspects that need to be kept in mind when interpreting and using CBA outcomes in the context of EU air policy. Each of these two aspects needs further consideration in the near future, and at presents limits the applicability of CBAs.

First, care needs to be taken in applying singular CBAs as a basis for policy setting. To ensure that policy development is robust, it is important that the policy does not focus on a single issue/value. For example, measures have been implemented over many decades to successfully reduce national pollutant emissions. Taking further reduction measures will soon result in diminishing returns and escalating costs. This implies that there may be more cost effective ways to achieve certain health or environmental benefits compared to reducing air pollution.

Second, the scenario's developed to support the air quality policy formulation process comprise a mix of health and environmental benefits (including particulate matter, ozone, eutrophying and acidifying substances). Only health benefits and ozone damage to crops are quantified in the monetary analyses. The costs and benefits of eutrophication and acidification control are not quantified and therefore not included in the CBA models constructed to date (AEA, 2011; EEA, 2011; EC4MACS, 2011). Nevertheless, ambitious targets for eutrophication and acidification control are included in the policy scenario's for air pollution control (A1-A6) presented in IIASA Report #10 and the 5 policy scenario's (LOW, low*, Mid, High* and HIGH) analysed in the aforementioned CBAs (gap closure compared to MTFR ranging from 25% in the LOW to 75% in the HIGH scenario's). The ambition levels for eutrophication and acidification have been established by association with the anticipated health benefits following PM emission reduction.

In order for CBA to properly play its role in informing the ambition setting process for meeting multiple targets in the most cost effective way (and provide transparency in the final impact assessment) it needs to be in a position to correctly attribute the incremental benefits and associated incremental costs for meeting each individual target. This is vital to ensure that benefits derived from achieving one target (e.g. PM health impact reduction) are not used to 'subsidise' the limited monetised benefits or lack of monetised benefits for meeting the additional target(s) (e.g. ozone health impact reduction). Therefore, eutrophication and acidification targets should be subject to a specific incremental CBA rather than be included in policy scenario's where the monetary benefits are driven by health impacts.

This need was highlighted by CONCAWE in its follow-up comments to SEG-4. However, in IIASA Report #10, the additional cost versus impact reductions in the step out scenarios from A3 (a high ambition PM only gap closure scenario) to A4-A6 does not develop such data but just asserts that the cost involved enables the capture of additional 'low hanging fruit'.

CONCAWE has undertaken a first assessment[33] of the additional marginal cost versus additional marginal benefits of the move from the A3 to A5 scenario using the CAFE approach for the valuation of ozone health benefits and an 'ecosystem services' approach to eutrophication and acidification.

As in CAFE, for ozone, the marginal costs exceed the marginal health benefits around a gap closure of 25%. Such a gap closure for ozone is already achieved as a 'come along' consequence of the high PM ambition scenario of A3 (see Figure 5.3 in Report #10). Any further incremental expenditure on reducing ozone impacts would therefore exceed the incremental benefit for this endpoint.

The expenditure to achieve the 25% ozone impacts gap closure is some 200M€/y; by subtracting this from the 894M€/y cost of A5 over A3, we obtain the necessary benefit figure for ecosystem services improvements (from reduced areas of eutrophication and acidification) required to justify the move from A3 to A5 scenarios, i.e. 694M€/y. The estimated (see footnote 2) increase in the area of ecosystem protected in moving from A3 to A5 is 3,500 km^2 for acidification and 3,500 km^2 for eutrophication. This would imply an improvement in ecosystem services value of about €1000/hectare/y to support the additional expenditure of 694M€/y. This is substantially higher than recent values for the average level of ecosystem services provided by European forests published in the literature (e.g. Matero et al., 2007; Zanderson et al., 2009; TEEB, 2010; Ding et al., 2010).

Without prejudice to earlier comments on the lack of justification for the A3 scenario, this clearly indicates the incremental costs in moving from A3 to A5 are also not supported by the incremental benefits.

Conclusions

This note shows that there are a number of points that should be addressed before the presently available CBAs of air quality targets can be used as a basis for robust policy formulation. In particular, there is an urgent need to use VOLY figures and monetised morbidity values that are in line with recent scientific insights, and to conduct proper uncertainty analysis to show the robustness of the cost benefit figures. In addition, there is a need to further examine alternative approaches to CBA for revealing the economic justification for policy setting and to conduct dedicated CBAs for the specific types of air pollutants (ozone, PM, eutrophying and acidifying substances).

[33] A more complete assessment can only be undertaken when the additional details on all the scenarios are released by IIASA; in particular scenarios A1 through A4.

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Attachment 1 (Technical Note)

An Assessment of the Valuation Methods for and Costs of Morbidity

June 2013

Contents

1 - Introduction

Recent Cost-Benefit Assessments (CBAs) on the health benefits of air pollution control, including those conducted for DG Environment are still grounded in the valuation approach developed in the CAFE project (AEAT, 2005). For instance, in the recent study 'Cost-benefit Analysis of Scenarios for Cost-Effective Emission Controls after 2020' (IIASA, 2012), the costs of morbidity are exclusively based on the CAFE estimates, adjusted for inflation. A concern is that many new studies on the costs of health impacts have become available in recent years, and that these are not being considered in recent CBAs on European air pollution control, such as the ones published by EEA (2011) and IIASA (2012). The costs of morbidity typically account for 20-25% of the costs of air pollution in Europe. Now that a number of recent publications of CONCAWE have examined the mortality costs of air pollution in Europe, and designed an enhanced, alternative method for determining the Value Of a statistical Life Year (VOLY), there is a need to consider the costs of morbidity.

This report reviews morbidity costs for the two most important morbidity effects of air pollution (in terms of financial costs, according to CAFE): chronic bronchitis and Restricted Activity Days (RADs). A review of the recent scientific literature is conducted to examine to what degree the monetised values for mortality and morbidity impacts applied in the recent CBA on air pollution control are aligned with recent scientific insights. However, these are given without prejudice to the quality and applicability of underlying health data, such as exposure response functions (ERFs), used to determine the health impacts of air pollutants in first place. The magnitude of these defined impacts again has a knock-on effect by influencing the size of their monetisation. CONCAWE has already provided its analysis of underlying health studies during the CAFE program. CONCAWE's main concerns with some of the health data are briefly summarised in Chapter 4 for completeness.

Chapter 2 analysis chronic bronchitis and Chapter 3 examines RADs. Since RADS are related to lost working days, the latter has been used in the policy arena to indicate that businesses would benefit from reducing air pollution, i.e. through a reduction in lost working days. The review included all studies available on-line in the published scientific literature that aimed to elicit the WTP for avoiding either chronic bronchitis and/or RADs in either a North American or European context. This number was, however, relatively low, only four studies analysing chronic bronchitis and two studies analysing RADs could be retrieved. Note that several more studies analysed the medical costs of chronic bronchitis (as presented in Table 1). The results of these studies are also included and serve as a reference for the total costs of chronic bronchitis (of which the medical costs are one of the elements, in addition to costs related to lost working days (which may be relatively low, see IGCB, 2005) and costs related to discomfort related to being sick).

Conclusions and recommendations are provided in Chapter 5.

2 - Chronic bronchitis

CAFE Volume 2 Report 'Health Impact Assessment' indicates that "there are few studies on air pollution and development of chronic bronchitis" and "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 CAFE cost figure for chronic bronchitis is based on only two studies: Viscusi et al (1991) and Krupnick and Cropper (1992). Both studies use a survey (CVM) approach, are undertaken in the US and were 15 years old at the time of preparation of CAFE (and over 20 years old at present).In addition, these studies use a definition of chronic bronchitis that does not coincide with those used in the epidemiological studies that attempt to quantify the number of cases due to air pollution. In particular, 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

The two studies used in CAFE indicate that this description constitutes the most severe definition of the chronic bronchitis endpoint. The Krupnick and Cropper study therefore attempts to scale these symptoms and compare an average and a severe case of chronic bronchitis. The authors estimate that the WTP for an average case of chronic bronchitis was 58% lower than that for the severe case.

Based on Viscusi et al (1991) and Krupnick and Cropper (1992) study, CAFE established the following estimate for the costs of chronic bronchitis (in euro):

High range estimate: € 265,692 (2003 prices, 250,000 in year 2000 prices)
Central range estimate: 200,000 (2003 prices, 190,000 in year 2000 prices)
Low range estimate: € 134,400 (2003 prices, 120,000 in year 2000 prices)

The central value, adjusted for inflation (208,000 euro) is used in IIASA (2012). However, the validity of using these values depends on the assumption 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. This case has not been made in the CAFE report, and the figures can be seen as indicative at best.

New studies. Since the research for CAFE was conducted, there have been a substantial number (>10) publications specifying the costs of chronic bronchitis, including in the EU (e.g. McGuire et al., 2001). These involved both studies examining the health costs associated with a case of chronic bronchitis and studies analysing WTP to avoid chronic bronchitis. A second point of critique therefore on the figures used at present is therefore that there is no consideration for the many recent studies published in this field.

To provide a first comparison, some alternative figures from the literature are summarised below. Wilson et al., 2000, estimate that the medical costs (in the US) related to chronic bronchitis amount to US$816 per year, including all costs of hospitalisation, medicines, visits to the physician etc. If for the sake of illustration it is assumed that these costs would be incurred over a 40 years period, the total medical costs are 40*816 = US$ 33,000 or around € 25,000 at current exchange rates. Note that these costs vary considerably as a function of the severity of the symptoms. Menn et al. (2012) found that the medical costs (visits to physicians, hospital treatments, medications) varied from less than € 20 euro to up to € 2812 per year in severe cases. Table 1 presents some of the values found in the literature.

Table 1. Comparison of the costs published on chronic obstructive pulmonary disease (COPD) in different countries. Note that COPD includes both the more dangerous emphysema and chronic bronchitis. Source: Chapman et al., 2006.

First author (ref.)	Country	Focus	Costs	Cost patient^{-1} yr^{-1}
Morera (52)	Spain	Top-down	Direct and indirect	€959
Hilleman (57)	USA	Bottom-up	Direct	Stage I US$1681 Stage II US$5037 Stage III US$10812
Jacobson (59)	Sweden	Top-down	Direct and indirect	
Wilson (60)	USA	Top-down	Direct	Emphysema US$1341 Chronic bronchitis US$816
Rutten van Mölken (56)	The Netherlands	Top-down	Direct	US$876
Dal Negro (56)	Italy	Bottom-up	Direct	Stage I €151 Stage II €3001 Stage III €3912
Jansson (54)	Sweden	Bottom-up	Direct and indirect	US$1284
Miravitlles (55)	Spain	Bottom-up	Direct	Stage I v1185 Stage II €1640 Stage III v2333
Masa (58)	Spain	Bottom-up Cross-sectional	Direct	€909.5

There are still relatively few studies on the WTP to avoid chronic bronchitis. O'Conor and Blomquist (1997) applied CVM to analyse WTP to avoid asthma in the US and found a value of around US$ 2200 per year. In a Swiss study, the contingent valuation method was applied to assess the reduction in quality of life due to chronic bronchitis. Interviewees - a representative sample of the general population - expressed their willingness-to-pay to reduce their risk of contracting the disease. Health was presented as a private good and respondents were made aware of the health implications, the average risk and the main causes of chronic bronchitis. The WTP to avoid chronic bronchitis was calculated to be CHF 38500 per case, at current exchange rates € 32,000 per case (Priez and Jeanrenaud, 1999). In Norway, Stavem (2002) undertook a detailed analysis of the WTP for a side-effect free cure for COPD (of which chronic bronchitis is the 'lighter' variant) among 59 COPD patients (average age: 57). The participants reported a median WTP of Norwegian kroner 200 000 (€ 25,000) for a theoretical cure for COPD without side-effects. This study also analysed the methodological aspects of applying CVM to diseases of cases such as chronic bronchitis and questions the methodological validity, showing among others that some of the estimates appear unrealistically high (which is why the study reports the median rather than the mean value) and noting that in Norway patients do not have to pay for their own medication.

Finally, in a European study covering 6 countries, the 6th Framework HEIMTSA study (Health and Environment Integrated Methodology and Toolbox for Scenario Development) analysed the WTP for avoiding several morbidity effects (Maca et al., 2011). Based on surveys in the Czech Republic, France, Germany, Greece, Norway, and the United Kingdom, they estimated the European central value for a morbidity case to be € 38,254 (Maca et al., 2011. This value is based on a mean value of the monthly WTP to avoid chronic bronchitis (see Table 2), with surveys being conducted in 2008 and 2009. The 95% confidence interval, however, does not reflect differences between countries which appeared to vary with a factor 2. The lowest value was found in the Czech Republic (mean WTP = € 183/month), and the highest in Greece (€ 606/month).

Table 2. Morbidity costs according to Maca et al. (2011).

Endpoint		Mean	99% C.I. (mean)	Median
Cough day	€ per case	25.6	23.8; 26.5	0
Chronic bronchitis	€ per month	318	304; 339	70
Mild COPD	€ per month	480	459; 505	140
Severe COPD	€ per month	734	698; 767	230
Asthma medication disomfort	€ per case	53.1	48.59; 59.34	19

Conclusions on chronic bronchitis. Based on this preliminary analysis, it can be concluded that the current figures used in the European CBAs on chronic bronchitis are too high. Already in the CAFE study it was noted that the studies presenting a WTP estimate to avoid chronic bronchitis (Viscusi et al. 1991 and Krupnick and Cropper 1992) use a definition of chronic bronchitis that is not in line with the epidemiological literature. In addition, several recent studies that consider both the costs of medication and the WTP to avoid the symptoms of chronic bronchitis result in much lower values. A better indication of the costs of chronic bronchitis in the EU is obtained by considering the Swiss and Norwegian studies, and by the study of Maca et al. (2011). If the average of the Swiss and Norwegian studies is taken, based on exchange rates, the WTP to avoid chronic bronchitis is around € 28,000, somewhat higher than the medical costs that are in the order of € 25,000. The value retrieved by Maca et al. (2011) was based upon a much larger number of respondents (around 11,000), in six European countries. The value was considerably higher, i.e. around € 38,000 in this study. Part of the difference arises from the fact that the Swiss and Norwegian values as presented in this report have not been corrected for inflation (this could explain some 5000 euro of the difference, assuming that the studies are 6 and 10 years older, respectively, and assuming an inflation (CPI) of 2.5% per year). Differences in the survey set-up and the description/severity of chronic bronchitis valued are also potential reasons for the deviation. A reasonable approximation of the value of chronic bronchitis may be in the order of € 28,000 to € 38,000 per case. This is approximately 1/6th to 1/7th the value used in the most recent European assessments of the costs of air pollution, such as IIASA (2012).

3 - Restricted activity days (RAD)

Introduction. According to the CAFE methodology, Restricted Activity Days (RADs) include: (i) days when a person needs to stay in bed; and (ii) days when a person stays off work. Together these days are labelled regular RADs. Days including other, less serious, restrictions on normal activity are called minor RADs, and are valued separately. They appear to correspond, to some degree, to a 'cough day' measured in some studies, however it needs to be noted that different studies use different definitions for regular and minor RADs.

In **CAFE**, The RADs are valued on the basis of aggregating the costs of foregone labour productivity and the costs of welfare loss. The costs of foregone labour productivity are estimated at 88 euro per day (median value, based on EUROSTAT and CBI data). Although it is not explained in CAFE, it is assumed that this is a per capita rather than a per employee value. Instead of the WTP to avoid a RAD, the WTP to avoid a minor RAD was taken, since it was assumed that the monetary value of the RAD available from Ready et al. (2004), see below, applies to a day spent sick in bed whereas a RAD also includes days spent off work but not sick in bed. The value of a minor RAD is 41 euro. CAFE Vol. 2 reports that a RAD is valued at 130 euro/day, and a minor RAD at 38 euro/day. CAFE also includes an adjusted value for a RAD, which is based on the notion that a person needs to take off from work but does not need to stay in bed. This day is valued at 83 euro per day. However, the figures do not add up, and presumably this estimate is based on the foregone productivity only. The values for a minor RAD and for the adjusted RAD are used by IIASA (2012) (however without the qualification 'adjusted'). In particular, the values used in the IIASA CBA study produced by M. Holland are 42 euro/day for a minor RAD and 92 euro per day for a RAD.

Ready et al., 2004. In order to verify the methodology presented by CAFE, the Ready et al. (2004) study was retrieved. Ready et al. (2004) define a bed day as "A day with flu-like symptoms including persistent phlegmy cough with occasional coughing fits, fever, headache and tiredness. Symptoms are serious enough that patient must stay home in bed for three days". A cough day was defined as "One day with persistent phlegmy cough, some tightness in the chest, and some breathing difficulties. Patient cannot engage in strenuous activity, but can work and do ordinary daily activities" – in line with the concept of a minor RAD.

The Ready et al. (2004) study interviewed, using CVM, approximately 1200 adults (slightly different for each of the six examined medical events), selected at random. Interviews were conducted, in each country, by a firm specialised in interviews based on a translation of the survey. The results of Ready et al. (2004) are presented in Table 3. Note that Ready et al. (2004) also examined the uncertainty level of their analyses, also in view of the unexpected result that the WTP is substantially higher in Spain and Portugal, where the average incomes are lower than in Norway, the UK or the Netherlands. The study indicates that benefit transfer can overestimate by as much as 230% or underestimate by as much as 77% (Ready et al., 2004).

Table 3. WTP to avoid a cough day, as retrieved in Ready et al. (2004) (euro).

	Pooled	Amsterdam	Oslo	Lisbon	Vigo	England
'Bed day'	49	36	60	44	56	42
'Cough-day'	41	42	54	42	57	30
Sample size	1138	174	196	109	391	268

Figures have been calculated from pounds into euro based on an exchange rate of 1.52, based on the prevalent exchange rate at the time of the survey (Ready et al., 2004).

Remarkably, the pooled WTP for a regular RAD is only 20% higher than for a minor RAD. What is even more remarkable is that in Amsterdam and in Vigo, Spain, the WTP to avoid a minor RAD exceeds the WTP to avoid a regular RAD. This points to the uncertainty levels involved in the results of Ready et al.

Comparison with other studies and data. Unfortunately, there are very few studies that provide alternative estimates of the WTP to avoid a RAD or a minor RAD. A rather old US study by Tolley et al. (1986) finds a value of US$52 for the WTP to avoid a RAD. Another way of getting an impression of the potential validity of the WTP figures for a RAD is to compare them with the VOLY figures found by Desaigues (and which exceed these of the Defra study, see the previous CONCAWE communications on this topic). Desaigues et al. (2011) find a VOLY of € 41,000 for the EU15 + Switzerland and € 33,000 for New Member Countries. The adjusted value of €92/day for an average 'adjusted' RAD, as used by IIASA, 2012 (stay home from work, may involve being confined to a bed, based on a temporary illness) adds up to € 34,000 for one year, exceeding the VOLY for the new member states (!).

For a minor RAD, which is valued at € 42 /day, if aggregated over a year a person would be willing to pay € 15,300 per year to avoid coughing a year round. It seems implausible that the value of a year round of minor RADs adds up to 45% of a VOLY for new member states and 38% for the EU15 and Switzerland. In Vigo, the figures found by Ready et al. (2004) and applied in the context of CAFE and the recent EU CBAs even seem to suggest that an annual aggregate of the WTP to avoid a cough day amounts to 20,000 euro (which compares to the Spanish GDP per capita of 23,000 euro). It is likely, however, that it is not defendable to extrapolate the WTP to avoid one sick day to larger time periods; this may reflect the same bias as found in the WTP for a VOLY where, on a per year basis, the VOLY becomes lower the longer the period for which the WTP is elicited (i.e. a VOLY established on the basis of a elicited WTP for a 3 months gain in life expectancy exceeds that of a VOLY based on an elicited WTP for a 6 months gain in life expectancy, ceteris paribus). Still, this effect has not been examined in relation to the WTP to avoid morbidity effects, and further clarification is needed.

A recently published study presents an additional benchmark to compare some of the Ready et al. (2004) figures with. Maca et al. (2011) examined the WTP to avoid several diseases including minor RADs (but not major RADs) in 6 European countries (this study was funded through the European Commission's 6th framework programme). Maca et al. (2011) found that the median willingness to pay to avoid a cough day, in 6 European countries, was € 0 per case; however, the mean value was € 26 per day. There were also considerable differences between countries, with the median value € 0 in Czech Republic, Germany, Norway and the UK, € 13 in Greece and € 15 in France. The mean value varied from € 12 in the UK to € 39 in Greece. Interestingly, the mean values found by the Maca et al. study (which included some 11,000 respondents) were about half the values found in Ready et al. (2004) (with a total sample size of 1138 respondents).

Conclusions on RAD. It is clear, as also acknowledged in Ready et al. (2004) (but not in CAFE or the follow-up CBAs) that there are important uncertainties in relation to the assumed value for a RAD. In particular:

1 - The value is based upon just one study (Ready et al., 2004)

2 - The results of this one study are internally inconsistent, specifically the outcomes that the WTP of Spanish and Portuguese to avoid a RAD far exceeds that of northern Europeans, and that the WTP to avoid a minor RAD exceeds that the WTP to avoid a 'regular' RAD in Amsterdam and Vigo. This may be related to a different phrasing or interpretation of CVM questions by the respondents, or the people holding the survey.

3 - Moreover, there is a methodological flaw in the CAFE approach. The WTP to avoid a sick day is added to the foregone labour productivity. However, it is likely that the WTP of respondents reflects in part the costs of foregone labour productivity, for instance in the case of a self-employed person, or in case a person fears for his or her job as a consequence of absence from work due to illness. These figures can therefore not be added one to one as is done in CAFE. However, it appears as if this latter mistake is avoided in the more recent CBAs (including IIASA, 2012), by taking the 'adjusted' value of a RAD, which corresponds to the foregone productivity due to absence from work. A question then is if RADs are equally distributed over people at working age and people below or above working age, which seems unlikely. It is unclear if this effect is compensated for.

4 - The values used in recent CBAs for a minor RAD are based on the Ready et al. (2004) estimate for a 'cough-day'. However these values are about a factor two higher than the values found in Maca et al. (2011) based on a much larger sample, and using more sophisticated statistical analysis.

Based on the available information it is very likely that the figures used in CAFE are significantly too high. However, since data are missing, it is difficult to say how much the costs of RAD are overestimated. Given that the median WTP in 4 out of 6 European countries for a 'cough day' is zero, it could be reasoned that a value of € 0 should be taken for a minor RAD. As for the value attributed to a RAD / bed day, it seems highly questionable if – pending further research - it is justifiable to use monetary figures with such a high degree of uncertainty as a basis for decision making.

4 - Quality and uncertainties of underlying health data

It is worthwhile stressing that in this report the focus is on the proposed methodology used for CBA. This methodology assumes the underlying relationship between exposure to PM2.5 and mortality and morbidity are causal. Concerning the morbidity effects of chronic bronchitis and restricted activity days (RADs), in CONCAWE's view, there are serious concerns with the causality assumption. Furthermore, in CONCAWE's view, the studies proposed for use in the CBA, Abbey et al. (1995) for bronchitis, and Ostro et al. (1987) and Ostro and Rothchild (1989) for RADs, are not of sufficient quality for use in a CBA. CONCAWE's scientific concerns with these studies are summarised below (for more detail see attached note on Concentration-Response Functions for Morbidity Endpoints under the Project HRAPIE)[1].

The study by Abbey et al. used the imprecise exposure metric of PM10 estimates derive from measurement of total suspended particulates and airport visibility data. The PM10 risk estimates have been converted to PM2.5 estimates for purposes of the CBA. Since bronchitis is primarily a disease of the upper respiratory tract, it is inappropriate to attribute bronchitis to PM2.5, a pollutant that distributes in the lower respiratory tract. The study was based on high levels of air pollution in California 20-30 years ago resulting in inflated exposure response functions (ERFs) that in our view are not applicable to evaluation of air pollution in Europe today. The results of the study are confounded by lack of control for smoking, a well-known risk factor for development of bronchitis. The results were also not significant at the 5% level. In our view, a single study reporting a non-statistically significant result should not be used in a CBA.

Similarly, in the studies proposed for use in evaluating RADs, PM2.5 levels were not measured. Rather, PM2.5 levels were estimated from PM10 measurements and visibility data from airports. The air pollution data evaluated were high levels in existence in California over 30 years ago are not applicable to Europe today. The health endpoint of RAD is highly subject to socioeconomic confounding. In the studies used to derive the ERFs, significant city to city differences in RAD rates were observed. This was likely due to socioeconomic factors and other factors that were not adequately controlled in the selected

[1] During the development of the CBA methodology for the CAFE program, CONCAWE provided detailed comments on these studies which have been so far not appropriately addressed. once again are proposed for use in an updated CBA.

studies. Some of these factors include time spent outdoors, building construction, health practices including how such days are recorded, age of the population, sex, race, education, income, marital status, temperature, employment conditions and rates, smoking rates, and many other factors. Even greater differences would be expected when extrapolating the results of these studies for use in Europe.

5 - Conclusions

This Technical Note has analysed the various approaches used in the scientific literature to value the costs of chronic bronchitis and Restricted Activity Days (RADs) and how these values are included in recent CBAs of European air quality policies. Together, these two effects are the most significant contributors to the morbidity costs in these CBAs, jointly accounting for some 20% of the costs of air pollution in Europe. This Technical Note shows that the scientific justification for the costs figures used for this part of the CBAs is very thin. The values are strongly overestimated, particularly in the case of chronic bronchitis. For chronic bronchitis, the IIASA (2012) study uses a value that is an estimated 6 to 7 times higher than recent and more scientifically robust estimates. In the case of RADs, the uncertainty in the results used in the CBAs is very high. Also, a more recent and thorough study (Maca et al., 2011), as compared to the study used in the recent CBAs (Ready et al., 2004), presents estimates for the costs of RAD that are around a factor 2 lower. This means that there is an urgent need to reconsider the values currently attributed to morbidity effects in the European CBAs on air pollution control.

Due to the concern on the quality of some of the underlying health data, the monetised values for chronic bronchitis and RADS established in this report have additional uncertainties attached to them than those discussed in this report and are most likely still conservative.

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Appendix 3: CONCAWE comments to the HRAPIE project

Concentration Response Functions for Morbidity Endpoints under the Project HRAPIE

A review of the concentration response functions for morbidity endpoints under the HRAPIE project was conducted. The purpose of this review was to provide a critique on the use of concentration response functions (CRFs), endpoints and its scientific relevance for use in the CBA. In addition, the current CBA analysis was compared to the scientific data used in the previous EU CAFE programme in the 2004 timeframe.

Endpoint: Bronchitis

The entire impact of the contribution of particulate matter on the incidence of bronchitis is inappropriately attributed to the fine particle fraction. This defies well-known biological facts concerning the etiology of bronchitis and highlights the need for clinical input into the CAFE CBA. It is very well known that bronchitis is primarily a disease of the upper respiratory tract. Therefore, coarse particles, which deposit in the upper respiratory tract, are much more likely to contribute to the etiology of this disease. Fine particles, which deposit in the lower respiratory tract, are not expected to contribute to the incidence of bronchitis. Therefore, it is not biologically appropriate to convert the morbidity function from a study using PM_{10} to $PM_{2.5}$. Rather, for bronchitis, a separate benefits analysis for PM_{10} or other coarse particle metric such as $PM_{2.5\text{-}10}$ or TSP should be provided. It is critical to note that in the study by Abbey et al, a stronger relationship was observed for TSP, the actual metric of particle exposure used, than for PM_{10} or $PM_{2.5}$. Therefore, valuation of bronchitis attributed to $PM_{2.5}$ should not be performed under the CAFE programe.

An appropriate method to convert the exposure response function (ERF) based on PM_{10} to an ERF based on $PM_{2.5}$ was not used in the CBA. The authors of the CBA take the attack rate for chronic bronchitis based on PM_{10} and adjust to get the results for $PM_{2.5}$. In our view, the attack rate should be adjusted to get the $PM_{2.5}$ fraction, and then taken out $PM_{2.5}$. Assume that 54% of the PM_{10} attack rate is due to $PM_{2.5}$. (A conversion factor of 1.54 is used). As such, 54% of the attack rate based on PM_{10} (which is 7%) becomes a 3.8% attack rate (54% x the 7%) PM_{10} attack rate = 3.8%). So, if 54% of the PM_{10} exposure is $PM_{2.5}$, this means a 10 ug/m^3 PM_{10} ERF is 5.4% $PM_{2.5}$. Final adjustment should be 5.4 ug/m^3 / 3.8% attack rate = 0.7% adjusted attack rate for $PM_{2.5}$. This compares with the authors adjusted rate of 1.07%.

The ERFs used were not for the air pollutant under consideration. Since monitoring of both PM_{10} and $PM_{2.5}$ was very limited in California before 1986, Abbey et al. used data for TSP to derive estimates for PM_{10} and airport visibility records to derive estimates for $PM_{2.5}$. This unwieldy approach to exposure estimate seriously jeopardized the findings from the study.

The assessment of bronchitis is based on a single study (ASHMOG Abbey et al., 1995a) for which the result was not even statistically significant at the 5% level. Causality cannot be established based on the results of a single ecological epidemiology study. Further, the accuracy of an ERF based on a single study result needs scientific justification, and in particular, the accuracy of the adjustment for smoking in this study, the major contributor to the incidence of bronchitis. The authors make the assumption that no smoking occurred in the cohort of seventh day Adventists. The higher lung cancer rates for males versus females in this study raises concern for this assumption. Further, 15% of the subjects in the study had smoked prior to 1977 and were then assumed to stop smoking when they became seventh day Adventists. Thirty percent of the study subjects lived with a smoker, and 42% had worked with a smoker. Further, the shear size of the risk due to PM air pollution, which is essentially the same size as the background rate attributed to all other factors, raises more suspicion. Finally, it is questionable to use an ERF based on a result that was not statistically significant at the 5% level. This brings into the question the concept of whether the findings from a single study are robust enough to conclude in the *primary or core* portion of a CBA. It is unlikely that such an approach would be normally justified and the CAFE CBA should apply a scientific process to accommodate and accept recommendations of the reviewers.

Using data from California during the period of 1966-1988 when air pollution was high, likely resulting in an inflated ERF. The authors of the CBA chose to partially justify inclusion of this endpoint based on reference to "modern" HIAs. The air pollution data that are the basis of the study used for the CBA are from 1966-1987, or close to 30 years old. It is questionable whether ERFs based on results from another continent using air pollution data from 30 years ago are sufficiently robust to use in a CBA designed to project results nearly 20 years into the future, a near 60 year extrapolation.

Using an ERF derived from high air pollution levels relevant to current, and with a different air pollution mix relative to those in Europe today requires further justification. Indeed, the air pollution data in California are dominated by photochemical smog. Likely, this ERF drastically over-estimates effects of low levels of PM alone. In fact, whether or not a threshold exists for this endpoint, and whether or not the ERF is specific to particulate matter, photochemical pollution, other gases present in ambient air, or a combination of these, has not been evaluated.

A baseline disease rate from a single U.S.-based study and extrapolation to Europe is used without any adjustment or consideration of the uncertainties. Only limited information is provided on how baseline rates in the U.S. compare with those in the U.S. It is well know that smoking is by far the major contributor to the production of bronchitis. One might guess therefore that the incidence of this disease might differ in the Europe versus the U.S. according to difference in smoking rates. Nonetheless, the accuracy of basing a benefits analysis for bronchitis based on baseline disease rates from the U.S is questionable.

Restricted Activity Days (RADS) and Minor Restricted Activity Days (MRADS)

Assessment of these endpoints are based on the results of a single study, the Health Interview Study, as reported by Ostro et al., 1987, and Ostro and Rothschild, 1989. The ERFs are derived from a study from another continent and during the period of 1976-1981, or close to 30 years ago, when the air pollution levels were higher. The validity of extrapolating from results in 1976 to 2020 and beyond, or close to a 50 year extrapolation is questioned. In the case of particulate matter, the tenuous exposure metrics used in this study is questioned and requires justification. PM_{10} and $PM_{2.5}$ levels were not measured as part of this study. Rather, $PM_{2.5}$ levels were estimated from visibility data from airports. Results of other CBA assessments have indicated concerns for extrapolating results from high pollution levels to lower levels and resulting inflation of the ERF at higher levels. Further, there has been no assessment of whether RADs or MRADs would even be triggered by lower air pollution levels. In other words, the issue of threshold has not been explored at all for these morbidity endpoints.

In the case of particulate matter, the adjustment of an ERF for PM_{10} to one based on $PM_{2.5}$ based on the simple mean ratio of these particles in urban air, and this practice is inappropriate. The authors offer no biological explanation as to why such an adjustment is appropriate, or why fine PM would be expected to exhibit the same potency as coarse particles. Fine and coarse particles distribute differentially in the respiratory tract and as stated by the WHO and others, produce a different and separate spectrum of health effects. As described above, certain respiratory symptoms would be expected to be exacerbated more by exposure to coarse rather than fine PM, a finding consistent with the actual study results reported by Abbey et al., where stronger associations were observed for TSP than for PM_{10} or $PM_{2.5}$ surrogates. It is not clear why the authors of the CBA choose to attribute all RAD related effects to fine PM.

In the case of ozone, the plausibility of the association with MRAD, the ERF selected and how it is applied in the CAFE CBA is questioned. Ozone is a respiratory toxicant. In the study by Abbey et al., no association was reported between exposure to ozone and *respiratory* restricted activity days (RRADs). This raises the question, if those in the study were not restricted due to respiratory-related reasons, what biologically related reason accounts for their restricted activity that could be due to ozone exposure? Using a multi-pollutant model applied to air pollution data between 1976 and 1981, the author's report a positive association between a 2-week average 1-hour ozone concentrations and MRAD. However, temperature is incorporated linearly in their mode and is highly correlated with ozone, which decreases the certainty that ozone alone is causing MRADs. In addition, there was high variance in the regression coefficients across the six years examined, with negative coefficients observed in 1977 and 1981 and a non-significant coefficient reported in 1976. Therefore, the conversion used by the authors of the CBA to convert an ERF based on 1-hr maximum levels to daily 8-hr averages should be properly justified. There is absolutely no question that higher peak concentrations of ozone produce more pulmonary effects than lower average levels. It is entirely possible that at lower average ozone levels, no respiratory effects and no MRADs would occur. However, the authors of the CBA did not consider this possibility and instead make the conversion to 8-hour average values, and extrapolate down to 35 ppb ozone, a level producing no clinical effects.

All effects of air pollution on RADs and MRADs are being arbitrarily attributed to fine PM and ozone, with potential effects of other pollutants ignored. For example, we note that in a multi-pollutant model, the hypothesized effect of exposure to $PM_{2.5}$ on RAD did not persist following adjustment for carbon monoxide (Steib et al, 2002).

The health endpoints of RAD and MRAD are highly subject to socioeconomic confounding. In the study used to derive the ERFs, significant city to city differences in RAD rates were observed. This was likely due to socioeconomic factors and other factors that were not adequately controlled in the selected study. Some of these factors include time spent outdoors, building construction, health practices including how such days are recorded, age of the population, sex, race, education, income, marital status, temperature, employment conditions and rates, smoking rates, and many other factors. Even greater differences would be expected when considering cities in the U.S. versus those in Europe. Further, many of the socioeconomic factors that need to be controlled to identify the potential effect of air pollution are likely much more important than air pollution itself in the production of RADs and MRADS. The analogy is attempting for a single drop of water inside an ocean, or a single ant within a colony of ants.

The RAD background rate taken from a U.S.-based study (ORNL/RFF, 1994) and inappropriately applied it to Europe's CBA. Socioeconomic factor such as disability rates, income status, unemployment rates, and various definitions of RAD will influence the background rates, and these factors were not considered.

In summary, all of the above indicate significant concerns for the transferability of these EFRs for use in assessing RAD and MRAD in "average Europe", without any consideration for all of the uncertainties involved. Certainly, such "benefits" should not be included in the core CAFE CBA assessment, either for particulate matter or for ozone. In the case of particulate matter, any estimates that are made should be attributed to coarse PM rather than fine PM.

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Appendix 4: Ecosystem services under the microscope

Ecosystem Approaches in support of Policy Formulation on Air Pollution, a Review

10 December 2013

Contents

1 - Introduction

Ecosystem approaches are defined for the purpose of this Paper as 'approaches to environmental management and policy making that aim to compare costs and benefits of management and policy options on the basis of an analysis of their impacts on the supply of benefits from ecosystems to people'. The benefits supplied by ecosystems to people have been labelled 'ecosystem services' (MA, 2003) and comprise such benefits as the provisioning of goods by ecosystems (e.g. wood, fish, genetic information), the regulation of environmental processes (e.g. water purification by wetlands, carbon sequestration in forests) and cultural services supplied by ecosystems (e.g. providing opportunities for recreation). The capacities of ecosystems to supply such services can be affected by air pollution and other types of environmental stress. Hence, ecosystem degradation, for instance through air pollution, evokes a cost expressed through a reduced supply of ecosystem services, and ecosystem rehabilitation through reduction of air pollution may lead to economic benefits through enhanced ecosystem services supply.

The Paper focuses on ecosystem approaches applied to the field of air emissions. The potential impact of air pollution is long range and, through reaction and transformation in the atmosphere, anthropogenic and biogenic emissions can combine to create pollutants having adverse effects on ecosystems and human health. Increasingly these items are coupled with events linked to climate change; for example changing precipitation patterns leads to crop losses and damages as does high ozone exposure. The combined effect of these processes may be more than a simple summation depending on plant functional responses.

The specific objectives of this Technical Concept Paper are:

- to conduct a screening of the state-of-the art of ecosystem approaches;
- to examine how ecosystem approaches are being used in support of on-going European policy formulation processes – particularly the Gothenburg Protocol and TSAP reviews;
- to analyse methodological gaps and uncertainties.

The paper is structured as follows. Chapter 2 provides an overview of the state-of-the art of ecosystem approaches to environmental management, including both ecological aspects and the application of environmental economics for the analysis of ecosystem impacts. Chapter 3 examines how ecosystem approaches are being used in support of on-going European policy formulation processes, in particular the Gothenburg Protocol and TSAP reviews. Chapter 4 analyses methodological gaps and uncertainties and proposes means of resolving these. Chapter 5 presents the key policy and scientific implications of this assessment. Annex 1 provides a Glossary of main terms used in the Paper.

2 - Brief overview of ecosystem approaches to environmental management

2.1 Ecosystem services

The UN Convention on Biological Diversity has provided the following definition of an ecosystem: 'A dynamic complex of plant, animal and micro-organism communities and non-living environment interacting as a functional unit'. Ecosystem services are the goods or services provided by the ecosystem to society (MA, 2003). Their supply depends on demand from society as well as the capacity of the local ecosystem to supply the service. For example, the amount of wood extracted from an ecosystem depends on the demand for wood and the costs at which wood can be obtained. The supply of ecosystem services will often be variable over time, and both actual and potential future supplies of services should be included in the consideration of ecosystem services in support of environmental policy making.

In the last two decades, ecosystem services have emerged as a central concept in environmental management due to its potential to link the physical and economic worlds. Key publications are the Millennium Ecosystem Assessment, which produced a framework for analysis in 2003 (MA, 2003) and a comprehensive analysis of ecosystem services globally in 2005 (MA, 2005), and more recently the publications of the TEEB (The Economics of Ecosystems and Biodiversity) Project (e.g. TEEB, 2010). In the scientific literature there are nowadays over a 1500 peer reviewed studies analysing ecosystem services supply in specific ecosystems, or providing methodological support to ecosystem services analysis and modelling. A new global assessment, IPBES (Intergovernmental Panel on Biodiversity and Ecosystem Services, analogous to the IPCC) is now in an early phase and planned to become the next global assessment in this field.

Four different categories of ecosystem services are distinguished in MA (2003), which may still be the most authorative global assessment: (i) provisioning services; (ii) regulation services; (iii) cultural services; and (iv) supporting services. These categories are described below, and Table 1 presents an overview of the ecosystem services in each category. Supporting services represent the ecological processes that underlie the functioning of the ecosystem. Their inclusion in valuation may lead to double counting as their value is reflected in the other three types of services, and this category is not further addressed in this Paper, in line with TEEB (2010) and CICES (2013).

(i) Provisioning services reflect goods and services *produced* by or in the ecosystem, for example a piece of fruit or a plant with pharmaceutical properties. The goods and services may be provided by natural, semi-natural and agricultural systems and, in the calculation of the value of the service, the relevant production and harvest costs have to be considered.

(ii) Regulating services result from the capacity of ecosystems to regulate climate, hydrological and bio-chemical cycles, earth surface processes, and a variety of biological processes. These services often have an important spatial aspect; e.g. the flood control service of an upper watershed forest is only relevant in the flood zone downstream of the forest. For instance, the nursery service is classified as a regulation service. It reflects that some ecosystems provide a particularly suitable location for reproduction and involves a regulating impact of an ecosystem on the populations of other ecosystems.

(iii) Cultural services relate to the non-material benefits people obtain from ecosystems through recreation, cognitive development, relaxation, and spiritual reflection. This may involve actual visits to the area, indirectly enjoying the ecosystem (e.g. through nature movies), or gaining satisfaction from the knowledge that an ecosystem containing important biodiversity or cultural monuments will be preserved. The latter may occur without having the intention of ever visiting the area. The cultural services category also includes the habitat service that represents the benefits that people obtain from the existence of biodiversity and nature (not because biodiversity provides a number of services, but because it is important in itself).

Table 1
List of ecosystem services (based on Turner et al., 2000; MA, 2003; TEEB, 2010)

Category	Examples of goods and services provided
Provisioning services	Food Fodder (including grass from pastures) Fuel (including wood and dung) Timber, fibres and other raw materials Biochemical and medicinal resources Genetic resources Ornamentals
Regulating services	Carbon sequestration Climate regulation through control of albedo, temperature and rainfall patterns Hydrological service: regulation of the timing and volume of river flows Protection against floods by coastal or riparian systems Control of erosion and sedimentation Nursery service: regulation of species reproduction Breakdown of excess nutrients and pollution Pollination Regulation of pests and pathogens Protection against storms Protection against noise and dust Biological nitrogen fixation (BNF)
Cultural services	Habitat service: provision of a habitat for wild plant and animal species Provision of cultural, historical and religious heritage (e.g. a historical landscape or a sacred forests) Scientific and educational information Opportunities for recreation and tourism Amenity service: provision of attractive housing and living conditions

There is an increasing interest in **species richness** as an indicator for ecosystem functioning and quality, in particular in the debate on air pollution impacts on eutrophication. The general reasoning here is that eutrophication can reduce species richness in terrestrial and aquatic ecosystems by changing the nutrient availability, favouring species better adapted to high nutrient availability over species better adapted to low nutrient availability. Species richness, in other words, in an indicator for the habitat service of ecosystems, reflecting the conservation value of an ecosystem. In addition, as made explicit in the Millennium Ecosystem Assessment (2003), biodiversity including species richness is a component required for the overall functioning of an ecosystem. In general, high biodiversity increases the resilience of the ecosystem against external shocks.

In the last decades, a large number of methods to quantify species richness and/or diversity have been developed. Three main categories of indicators for species richness are briefly described below:

- **Number of species in specific classes.** Indicators presenting the species richness of an area often focus on (a combination of) specific taxonomic groups, such as mammals, meadow birds, or vascular plants. Although the number of species in specific groups is an indicator of the species diversity of an area, drawbacks are that it does not indicate the population numbers per species (which may or may not be below viable population numbers) and that it gives equal weighing to each species.

- **Biodiversity indices.** The most well-known of these indicators are the Simpson and Shannon Indices. They express the species diversity in an ecosystem, taking into account both species richness and the relative abundance of each species. However, the indicators are difficult to interpret and require a lot of data on species occurrence. In addition, they provide equal weighing to each species (attributing equal value, for instance, to a conservation flagship species and a pest).

- **Numbers of red-list and/or endemic species.** The IUCN Red List has a global cover and provides taxonomic, conservation status and distribution information on plants and animals. The number of species evaluated for the list is currently (2009) over 45,000. Certain taxonomic groups have been comprehensively assessed (e.g. mammals, birds, amphibians, freshwater crabs, warm-water reef building corals and conifers). The cover is not complete for all taxonomic groups, with data deficiencies remaining for freshwater, marine and semi-arid ecosystems. The list provides a good starting point for identifying the number of species of particular concern for nature conservation that are present in an ecosystem.

2.2 Economic valuation of ecosystem services

Valuation of ecosystem services involves a number of subsequent steps, i.e. (i) definition of the boundaries of the (eco) system and identification of the services to be studied; (ii) quantification of ecosystem services in biophysical terms; (iii) valuation of ecosystem services; and (iv) aggregation or comparison of values of different services. The services to be in or excluded from the assessment are determined by the objectives and system boundaries of the assessment. For regulating services biophysical quantification may require detailed modelling, often spatially explicit (i.e. in a GIS), of the various relevant ecological and biochemical processes in an ecosystem. Cultural services are strongly dependent on the cultural backgrounds of the people that receive the service and may depend on religious, moral, ethical and aesthetical motives. The most tangible cultural service, that can be analysed by means of a Travel Cost Valuation Method, is recreation and tourism. Other services, in particular those supplying non-use benefits (see below) are much harder to quantify and the related uncertainties in the valuation are usually substantial.

Ecosystem services can provide different types of economic value. In literature, four types of economic value are often distinguished, even though different authors have provided different classifications for these value types (e.g. Pearce and Turner, 1990; Munasinghe and Schwab, 1993; MA, 2003). Generally, the following four types are recognized: (i) direct use value; (ii) indirect use value; (iii) option value; and (iv) non-use value.

(i) Direct use value arises from the direct utilisation of ecosystems, for example through the sale or consumption of a piece of fruit. All provisioning services and some cultural services (such as recreation) have direct use value.

(ii) Indirect use value stems from the indirect utilization of ecosystems, in particular through the positive externalities that ecosystems provide. This reflects the type of benefits that regulation services provide to society.

(iii) Option value relates to risk. Because people are unsure about their future demand for a service, they are willing to pay to keep the option of using a resource in the future – insofar as they are, to some extent, risk averse. Option values may be attributed to all services supplied by an ecosystem.

(iv) Non-use value is derived from attributes inherent to the ecosystem itself. There are three types of non-use value: existence value (based on utility derived from knowing that something exists), altruistic value (based on utility derived from knowing that somebody else benefits) and bequest value (based on utility gained from future improvements in the well-being of one's descendants).

These four value types all need to be considered in the assessment of the total value of the services supplied by an ecosystem. In principle, the values are additive. Insofar as commensurable value indicators have been used, they may be summed in order to obtain the total value of the services supplied by the ecosystem. However, when analysing the ecological economics literature, there is relatively little experience with valuation of option values. The valuation of non-use values, which can only be done with stated preference methods such as Contingent Valuation Methods, is also prone to significant uncertainty. A brief overview of valuation methods is presented below.

Valuation of private goods. In the case of private goods or services traded in the market, price is the measure of marginal willingness to pay for that good, under perfect market conditions. Valuation of changes in the supply of a market ecosystem service, for instance as a function of changes in air pollution levels, requires establishing changes in the consumer and the producer surplus generated by the service. This generally requires analysing the demand and the supply curve for the ecosystem services in question. In case of price distortions, for example because of subsidies, taxes, etc., an economic (shadow) price of the good or service in question needs to be constructed. In some cases, this can be done on the basis of the world market prices following well-established approaches (Little and Scott, 1976). In case the private good is not traded in the market, because it is bartered or used for auto-consumption, shadow prices can be constructed, for instance on the basis of: (i) the costs of substitutes; or (ii) the derived benefit of the good (Munasinghe and Schwab, 1993).

Valuation of public goods. For public goods or services, the marginal willingness to pay cannot be estimated from direct observation of transactions, and the demand curves are usually difficult to construct. Two types of approaches have been developed to obtain information about the value of public ecosystem services: the revealed and the stated preference approach (Pearce and Howarth, 2000). The revealed preference approaches use a link with a marketed good or service to indicate the willingness-to-pay for the service. There are two main types of revealed preference approaches:

- *Physical linkages.* Estimates of the values of ecosystem services are obtained by determining a physical relationship between the service and something that can be measured in the market place. The main approach in this category is the damage-function (or dose-response) approach, in which the damages resulting from the reduced availability of an ecosystem service are used as an indication of the value of the service. This method can be applied to value, for instance, the hydrological service of an ecosystem.

- *Behavioural linkages*. In this case, the value of an ecosystem service is derived from linking the service to human behaviour – in particular people's expenditures to offset the lack of a service, or to obtain a service. An example of a behavioural method is the Averting Behaviour Method (ABM). There are various kinds of averting behaviour for instance defensive expenditure (a water filter) or the purchase of environmental surrogates (bottled water). The travel cost method and the hedonic pricing method are other indirect approaches using behavioural linkages.

With *stated preference* approaches, various types of questionnaires are used to reveal the willingness-to-pay of consumers for a certain ecosystem service. The most important approaches are the Contingent Valuation Method (CVM), Choice Experiments and related methods. In the last decades, CVM studies have been widely applied. It is the only valuation method that can be used to quantify the non-use values – for instance those related to biodiversity conservation - of an ecosystem in monetary terms. Various authors question their validity and reliability - both on theoretical and empirical grounds. There are two main points of criticism against CVM. First, CV estimates are sensitive to the order in which goods are valued; the sum of the values obtained for the individual components of an ecosystem is often much higher than the stated willingness-to-pay for the ecosystem as a whole. Second, CV often appears to overestimates economic values because respondents do not actually have to pay the amount they express to be willing to pay for a service (see e.g. Cummings and Harrison, 1995; Hanemann, 1995).

2.3 Ecosystem dynamics

Since some three decades, ecologists have become aware that ecosystem change is in the majority of the cases determined by complex, non-linear dynamics rather than linear responses to management and stress. These dynamics are critically important for ecosystem approaches, also to air pollution management, because they determine the response of an ecosystem to either an increase or a reduction in pollution loads. In addition, the supply of ecosystem services is often directly linked to the state of the ecosystem. Ecological research on complex dynamics is still continuous, even though the theory is now well established and increasingly integrated in environmental policy making. Note that also the RAINS (Regional Air Pollution Information and Simulation) model used for modelling critical loads in relation to air pollution is grounded in the concept of complex dynamics, as elaborated below.

In short, complex dynamics are irreversible and/or non-linear changes in the ecosystem as a response to ecological or human drivers. Below, the following key elements of complex ecosystem dynamics are briefly discussed: (i) irreversibilities; (ii) multiple states and thresholds; and (iii) stochasticity and lag-effects.

(i) Irreversible dynamics. Irreversible changes in ecosystems occur when the ecosystem is not, by itself, able to recover to its original state following a certain disturbance. Irreversible changes may be permanent, as in the global loss of a species, or they may only be reversed through substantial interventions in the ecosystem, for example in the case of reforestation on sites where natural processes would not lead to recovery of the tree cover. Irreversibility comprises different mechanisms, and can take place at different scales. For instance, it can relate to the extinction of a particular species, or the conversion of an ecosystem. It may also refer to irreversible changes in the state of an ecosystem, as in the case of a transition from a rangeland dominated by palatable grasses to one dominated by unpalatable shrubs. At the global scale, the increased loading of the atmosphere with carbon dioxide is an example of a process that can be considered as irreversible at human time scales. Irreversible change may either be rapid, involving a threshold, or more gradual. Often, it is subject to considerable uncertainty, for instance with reference to the location of the threshold, or the overall rate of change of the system following a disturbance (e.g. Scheffer and Carpenter, 2003). Note that in case of irreversible dynamics, reduction in pollution loading does not yield economic benefits since recovery of the ecosystem does not take place.

(ii) Multiple states and thresholds. Multiple states are relatively stable configurations of the ecosystem, caused by the existence of feedback mechanisms that reinforce the system to be in a particular state (Scheffer et al., 2001). The state of the ecosystem may be a consequence of physical or biological perturbation, such as changes in nutrient loading or species deletion or invasion. The probability that a disturbance leads to a shift from one state to the next depends upon the magnitude of the disturbance and on the resilience of the current state. Often, the shift between multiple states occurs suddenly and comprises the existence of threshold effects. Multiple states and thresholds have been observed in a range of ecosystems, including freshwater lakes, marine fish stocks, woodlands, rangelands, coral reefs and coastal estuaries.

A type of dynamics that occurs, in some ecosystems, in conjunction with multiple states and thresholds is hysteresis. Hysteresis occurs when the ecosystem's response to an increasing pressure follows a different trajectory from a response to a release in pressure. An example is provided by the response of an estuary to nutrient loading. At low nutrient loads, seagrass may dominate the flora, but with increased nutrient loading the phytoplankton concentrations gradually increase. At a critical load the phytoplankton concentration is so high that seagrass does not have enough light to grow. The seagrass population collapses, which allows the phytoplankton to grow to even higher concentrations. To re-establish the seagrass beds, nutrient loads have to be reduced considerably below the critical load. Other ecosystems in which hysteresis has been detected include shallow lakes, rangelands, hemlock-hardwood forests and deep lakes.

(iii) Stochasticity and lag-effects. The ecosystem may also develop as a consequence of stochastic natural conditions, for instance when ecosystem change is driven by fires or high rainfall events. In the marine environment, major changes in the dominant fish species occupying a particular niche may be triggered by relatively minor, stochastic fluctuations in the fish community. Lag effects appear when impacts of specific drivers occur with a certain delay, for example because changes need to be triggered by a specific event. For instance, in rangelands, the impact of soil degradation resulting in reduced seedling establishment may become apparent only after a fire.

Hence, complex dynamics are of major importance for the understanding of ecosystem dynamics. They determine the response of the ecosystem to management including changes in air pollutant concentration. Since the capacity of an ecosystem to supply ecosystem services depends on the state of the ecosystem, the application of ecosystem approaches to air pollution control needs to consider these complex dynamics. A summary of complex dynamics is provided in Figure 1. The left hand graph presents the traditional, but seldom applicable case of a gradual, reversible response of an ecosystem to stress (e.g. stress from air pollution). The other two graphs present different types of complex dynamics. The RAINS model uses the concept of critical loads, which is a representation of the occurrence of thresholds in soils at which a rapid decrease in pH can be expected (due to the occurrence of Calcium and Aluminium buffers), as presented in the middle graph. When the critical load (stress level) is exceeded, a rapid change in pH of the forest soil can be expected, with subsequent consequences for the vitality of the forest and the supply of ecosystem services (e.g. timber production, tourism). The right hand graph pictures an ecosystem subject to irreversible dynamics.

In relation to air pollution control, there is an increasing interest in dynamic modelling of the impacts of acid and eutrophying substances on ecosystems. Dynamic modelling provides insights in the response of ecosystems, over time, to reductions or increases in pollution loads and allows a more accurate analysis of costs and benefits of air pollution control, once these models have been developed with sufficient degree of reliability (see e.g. Hettelingh et al., 2007 for an overview) and provided that adequate monetary valuation of changes in ecosystems can be achieved.

3 - Ecosystem Approaches in Air Quality Policies

3.1 The Gothenburg Protocol

The 1999 Gothenburg (Multi-effect) Protocol is part of the Convention on Long-Range Transboundary Air Pollution (CLRTAP). The CLRTAP includes eight protocols that identify specific obligations to be taken up by the Parties and has been signed by, at present, 32 countries including most western European countries, Canada, the Russian Federation, Ukraine and the USA. The Gothenburg Protocol was signed in 1999 in Gothenburg and entered into force in 2005. It sets emissions ceilings for sulphur dioxide, nitrogen oxides, volatile organic compounds and ammonia in order to reduce acidification, eutrophication and ground-level ozone.

The Annexes of the Protocol allow Canada and the USA to participate with different commitments than other signatory parties. This is due to the different regulatory nature of Canada and the USA versus most European countries. In the EU, the Gothenburg Protocol has been implemented through the National Emission Ceilings (NEC) directive. The NEC directive is more recent than the Gothenburg Protocol and deviates slightly from it. Key environmental standards specified in the Gothenburg Protocol are listed below:

- Critical loads and levels
- Maximum allowable emissions (emission ceilings) for sulphur, nitrogen oxides (NOx), Volatile Organic Compounds (VOCs) and ammonia (NH_3).
- Emission limits for sulphur from stationary sources
- Emission limits for nitrogen oxides (NOx) from stationary sources
- Emission limits for Volatile Organic Compounds (VOCs) from stationary sources
- Emission limits for fuels and new mobile sources
- Emission limits for ammonia (NH_3) from agricultural sources

In the Gothenburg Protocol, emission limits are set for each participating country. The emission limits were negotiated on the basis of scientific assessments of pollution effects and abatement options. The selection of the specific emission levels was based on the predicted effects of the pollutants and the costs of controlling pollution. The Protocol also sets limit values for specific emission sources (e.g. combustion plants, electricity production, dry cleaning, cars and lorries) and prescribes best available techniques to be used for specific applications. Hence, the use of cost benefit approaches to establish emission targets is fundamental to the Protocol.

Central in providing the science behind the Gothenburg Protocol was the RAINS model. This model links sectoral developments and abatement measures for various pollutants with environmental impacts of air pollution. RAINS covers acidification, eutrophication, ozone damage to vegetation, and health effects due to exposure to ozone and primary and secondary particulate matter. In 2007 the RAINS model has been extended into the GAINS (Greenhouse Gas and Air Pollution Interactions and Synergies) model that also includes greenhouse gas emissions and structural measures that affect the activity levels.

Substantial amendments to the Gothenburg Protocol were agreed in May 2012. These amendments included new commitments for the reduction of $PM_{2.5}$, specific attention for black carbon as driver for both air pollution and climate change, and new commitments to reduce the emissions of sulphur dioxide, nitrogen oxides, ammonia, and volatile organic compounds. In addition, a number of new countries signed up for the Gothenburg Protocol. Significant improvements in air quality can be expected as result of the implementation of the revised Gothenburg Protocol.

Major reductions in acidification have been achieved in Europe, in particular as result of the reduction of sulphur dioxide emissions. Based on updated data on critical loads for acidification and eutrophication for Europe, it was estimated that critical loads for acidification will be exceeded at 11 percent of the European ecosystem area in 2020, compared to 34 percent in 1990 and 20 percent in 2000 (CIAM, 2007). Nitrogen deposition, from a wide variety of sources including agriculture, will however still exceed critical loads for eutrophication in 53 percent of the ecosystem area (CIAM, 2007). There is, to date, still considerable uncertainty on how these emissions and subsequent changes in ecosystems changes have affected ecosystem services supply in Europe. The aspect that has received most attention is modelling ozone damages, in particular to crops. CIAM (2007) refers to the UK-based International Cooperative Programme on Effects of Air Pollution on Natural Vegetation and Crops (ICP), that is able to detect ozone damages across 17 European countries, and which has revealed damages of ozone in, in particular, South Germany and the Mediterranean.

The CIAM (2007) review of the Gothenburg Protocol indicated that the Protocol bases emission reduction targets on the principle of critical loads and thresholds rather than on an ecosystem approach where the benefits of reducing pollution (due to an enhanced supply of ecosystem services) are compared with the costs (in terms of pollution control measures). Nevertheless, several preliminary figures are mentioned in the review, including damage costs for ozone. No costs for nitrogen deposition were specified. The updated CIAM (2011) study analyses the cost-effectiveness of various emission reductions scenario's to improve air quality in Europe in 2020 but does not present a further specification of the economic benefits resulting from reduced pollution in ecosystems.

3.2 The Thematic Strategy on Air Pollution

The Thematic Strategy on Air Pollution (TSAP) (September 2005) supplements national and preceding EU legislation by establishing objectives for air pollution and proposing measures for achieving them by 2020. The Strategy on Air Pollution is one of the seven thematic strategies provided for in the Sixth Environmental Action Programme adopted in 2002. It is the first of these strategies to be adopted formally by the Commission. It is based on research carried under by the Clean Air For Europe (CAFE) programme and the successive research framework programmes, and was adopted following a lengthy consultation process involving the European Parliament, Non-Governmental Organisations and industry and private individuals.

The TSAP covers a wide range of air quality issues and potential pollutants, with a focus on Particulate Matter. The TSAP sets health and environmental objectives and emission reduction targets for selected key pollutants. Emission reduction objectives will be delivered in stages, and aim to reduce particulate matter and ozone concentrations in air, with associated impacts on acid rain, excess nutrient nitrogen and ozone. The generation of benefits through reduced exposure of ecosystems to air pollutants is one of the drivers behind the TSAP.

Concerning ecosystem approaches, the EU legislative page states that 'there is no agreed way to assign a monetary value to ecosystem damage or the likely benefits resulting from the Strategy'. However, it is stated that 'there should be a favourable impact as a result of reducing acid rain and nutrient nitrogen inputs, resulting among other things in better protection for biodiversity'. At the same time, the Commission is pursuing work aimed at quantifying benefits from reduced exposure of ecosystems to air pollution.

In particular, the Commission concluded in 2007, with Consultancy Services from ARCADIS Ecolas, a road map for enabling the monetary assessment of ecosystem benefits of air pollution abatement policies (De Smet et al., 2007). This study reviewed studies aimed at valuing benefits of ecosystem responses to reduced air pollution, and found the following constraints:

1 - Reduction scenarios in existing studies do not match those of current European policy initiatives
2 - The number of studies is limited, and there is incomplete coverage of ecosystems and services. In particular there is a lack of studies that link (reductions in) air pollution to ecosystem services and economic benefits.
3 - The ecology of the studies was often inadequate
4 - Many dose response relations are quite uncertain
5 - There is lack of information on non-use values attributed to ecosystem services.

In terms of a pathway towards defining ecosystem approaches in support of policy making on air pollution control, the study recommends to carry out a number of case studies in representative and major EU ecosystems, in which emissions are linked to concentrations and subsequently to services and economic value of changes in services supply.

Since 2006, there still have been few studies that explicitly link air pollution to ecosystem benefits in the scientific literature (e.g. Bytnerowicz et al., 2007, see also Section 4), and it is likely that these on-going scientific advances will be reflected in the further TSAP discussions, given the stated need (De Smet, 2007) to also include ecosystem impacts in the policy formulation process.

4 - Methodological gaps and uncertainties

4.1 Overview of the assessment methodology

Applying an ecosystem services approach to air pollution policy involves expressing ecosystem impacts in monetary terms, and comparing the costs of pollution control measures to the benefits of reduced pollution loads in ecosystems. This approach is consistent with and complementary to the use of environmental cost-benefit analysis to quantify health impacts in monetary terms in support of policy making. In principle, benefits from enhanced health impacts and from enhanced supply of ecosystem services can be added in the calculation of the social welfare generated by reduced pollution in the overall environment. An important difference is that ecosystem valuation does not require the translation of health impact and mortality in monetary terms, an item that is still controversial and prone to considerable uncertainty.

The next sections of this chapter discuss the key methodological gaps and uncertainties in relation to three key steps in the assessment process that are in particular prone to uncertainty. These three key steps are: (i) establishing ecological dose-response relationship linking emissions to impacts on ecosystems; (ii) linking ecosystem change to changes in ecosystem services supply, and (iii) economic valuation of ecosystem services. The current status with regard to modelling and analysing these aspects in relation to air pollution, as well as key methodological gaps and uncertainties are analysed below. The sections will also briefly explore pathways to address the identified methodological gaps and uncertainties.

4.2 Uncertainties in ecological modelling of dose-response functions

Nitrogen loading and ozone exposure cause changes in plant chemistry, photosynthesis, and ecosystem carbon balance in sensitive ecosystems. As transport and deposition of emissions continues, high N loading and air pollution (especially ozone exposure) may produce similar changes in less sensitive systems. Additional responses at these and larger scales may include shifts in dominant plant species, export of nitrates and acidity to streams, rivers, and estuaries, coastal eutrophication and harmful algal blooms and, possibly, increased invasiveness by N-demanding species (Grimm et al., 2008). Nevertheless, there remain a number of uncertainties in relation to ecological dose response relationships. This section provides a brief overview of challenges and key uncertainties in relation to modelling interactions between air pollutants and ecosystems in the natural environment.

The most typical approach to documenting the effects of specific pollutants is a dose-response experiment, where the objective is to develop a regression equation describing the relationship between exposure and some easily measured effect (e.g., growth, yield or mortality). As analytic methods improved and ecology progressed, a broader range of effects of air pollutants is now identified and the understanding of the mechanisms of effects improved. Observations made on various temporal scales (e.g., long-term studies) and spatial scales (e.g., watershed studies) led to the recognition that air pollution can affect all organizational levels of biological systems including individuals, communities, species, and the ecosystem.

Several general points emerge from a review of ecological effects. First, air pollutants have indirect effects that are at least as important as direct toxic effects on living organisms. Indirect effects include those in which the pollutant alters the physical or chemical environment (e.g., soil properties), the plant's ability to compete for limited resources (e.g., water, light), or the plant's ability to withstand pests or pathogens. Examples are excessive availability of nitrogen, depletion of nutrient cations in the soil by acid deposition, mobilization of toxic elements such as aluminium, and changes in winter hardiness. As it is true for other complex interactions, indirect effects are more difficult to observe than direct toxic relationships between air pollutants and biota, and there may be a variety of interactions that have not yet been detected.

Damages to ecosystems are often caused by a combination of environmental stress factors. These include anthropogenic factors such as air pollution and other environmental stress factors such as low temperature, excess or limited water, and limited availability of nutrients. The specific combinations of factors differ among regions and ecosystems where declines have been observed. In addition, there is a group of substances that can be conserved in the landscape after they have been deposited in ecosystems. These substances are transformed through biotic and abiotic processes and can accumulate in the ecosystem. They include hydrogen ions ($H+$), sulphur (S), nitrogen (N), and mercury (Hg). Deposition of these pollutants can result in progressive increases in concentrations and affect ecosystems due to cumulative effects. Their effects can also continue after the stressors themselves have been reduced.

Pollutant-environment interactions are further complicated by the fact that biotic and abiotic factors in ecosystems change significantly over time due to ecological processes. Besides oscillations on a daily basis, and changes in a seasonal rhythm, there are long-range successional developments over time periods of decades. Table 2 presents an overview of key uncertainties in relation to establishing dose-response relations for air pollution impacts on ecosystems.

Table 2
Key uncertainties in dose-response relations for air pollution impacts on ecosystems
Source: EPA, 1999; CIAM, 2007; Grimms et al., 2008, Smart et al., 2011.

Uncertainty	Comment
Multiple pollutants	Pollutants interact at the ecosystem level and may enhance or alter impacts depending on the ecosystem type and the pollutants involved.
Multiple stressors	Some pollutants may reduce the resilience of the ecosystem to other pollutants or other types of stress including stress resulting from climate or land use change.
Ecosystem responses and adaptive capacities	Ecosystems poses complex dynamics where impacts in terms of pollution loading may trigger a range of positive and negative feedback mechanisms.
Ozone flux modelling	Flux-based approaches consider the uptake of O_3 by plants depending on humidity and other conditions but further work is needed.
Impacts of forests on air quality	Forest absorb $PM_{2.5}$ but there are few measurements of absorption rates in European ecosystems, and it is as yet unclear at what scale these impacts occur.

4.3 Uncertainties in linking ecosystem change to ecosystem services supply

Once specific changes in ecosystem state (e.g. changes in NPP, species composition, etc., as discussed in Section 4.2) have been modelled, they need to be linked to changes in the physical supply of ecosystem services. In this next step, there are a number of important additional methodological gaps and sources of uncertainty, as discussed in this section. These methodological gaps differ for crops and for (semi-) natural ecosystems such as forests, given that croplands are much simpler in terms of vegetation composition and structure. A brief overview of methodological gaps and uncertainties is presented below, specific for croplands and (semi-)natural ecosystems.

Croplands. Pollutants may affect processes within plants that control or alter growth and reproduction, and affect yields. Potential impacts include decreased photosynthesis, changes in carbohydrate allocation, increased foliar leaching, and increased sensitivity to stress. Air pollutants that may damage plants include O_3, SO_2, NOx and VOC. These pollutants may have direct effects on crops or may damage crops indirectly by contributing to ground level O_3 concentrations and/or acid deposition. While all of the above air pollutants may inflict stress on plants and affect crop yields, in most cases pollutants other than ozone are not a significant danger to crops (EPA, 1999). In addition, N deposition may enhance crop yields through a fertilisation effect (note that in general wet and dry deposition rates of N are substantially lower than N application rates through manure and inorganic fertilisers in most intensively managed fields).

Other ecosystem types. The work on air pollution impacts on other ecosystems has focussed on forests and water bodies. Available studies indicate the type of effects but dose response relationships have for most service not been established, the only exception being timber production from forests for which tentative data appears to be available (CIAM, 2007). The dose response relations for other ecosystems are substantially more complex than for cropland due to a number of factors: (i) the large diversity of ecosystems and their soil types, vegetation, etc.; (ii) the diversity of services provided by ecosystems; and (iii) in addition to ozone, some ecosystems are also significantly affected by SO_2 and nitrogen deposition. Table 3 provides an overview of the current information available on the impacts of air pollution on ecosystem services supply.

Table 3
Air pollution impacts on ecosystem services substantiated in the literature.
Adapted from EPA (1999), Bytnerowicz et al. (2007), Grimm et al. (2008) and Smart et al. (2011).

Ecosystem service	Impacts
Provisioning services	
Timber production	Loss of timber production due to ozone damage and acidification. Positive effect of nitrogen fertilisation can occur in nitrogen limited ecosystems.
Fish production	Acidification may lead to reduced fish production in lakes. Effects of eutrophication on fish populations may be positive (productivity) or negative (algal blooms, changes in species composition)
Regulating services	
Carbon sequestration	Positive impacts from nitrogen deposition in nitrogen limited ecosystems, negative impacts from ozone pollution and acidification.
Cultural services	
Recreation	Reduced recreation in ecosystems visually affected by acidification or eutrophication, reduced opportunities for recreational fishing (e.g. salmon fishing) in acidified freshwater
Biodiversity conservation	Changes in biodiversity and rare species throughout Europe due to eutrophication and acidification.

It is clear that there are considerable uncertainties related to linking air pollution to impacts on the physical supply of ecosystem services, in particular for other services than crop production. In general, such relations can only be established with elaborate models incorporating drivers for ecosystem change, key ecosystem state variables, and services supply. Contrary to effects on crop yields, it is generally very difficult to establish such relations with experiments because it is next to impossible to expose an overall ecosystem to controlled pollutant concentrations and because analysing ecosystem service supply under different pollutant concentrations based on observations faces the challenge of singling out the impact of the pollutant in between a myriad of other factors driving ecosystem services supply.

Another important factor is that ecosystem changes, for instance effects of acidification are subject to complex dynamics. Non-linear responses including a strong response once a threshold is passed can be expected for impacts of acidification and eutrophication (Scheffer et al., 2001) and potentially in ecosystems also in relation to ozone exposure[34]. In general, for many ecosystems, based on currently available information, it appears as if ecosystem services supply is strongly related to ecosystem state rather than the amount of pressure exerted on an ecosystem. In other words, the deposition of acidifying or eutrophying substances in an ecosystem only has a strong impact on the supply of ecosystem services once a threshold is passed (e.g. a buffer is exceeded) and the ecosystem shifts to an alternative state. This further complicates the analysis of ecosystem impacts, since the role of thresholds is crucial, and there remains uncertainty on the precise pressure at which the threshold is exceeded.

Hence, the key methodological gaps and sources of uncertainty in establishing the relation between ecosystem change and services supply are as follows. First, there are insufficient studies to establish general dose response relations, even for those ecosystem services for which there are concrete studies available. Second, changes in ecosystem services are guided by the complex dynamics of ecosystems, and there is insufficient information on thresholds determining ecosystem responses to acidification and eutrophication. Third, there is insufficient insight in how the effects of different pollutants may influence one another, and how these effects may interact with other pressures on ecosystems, for instance from climate change. These key sources of uncertainty need to be addressed in further scientific assessments before an ecosystem services approach can be quantitatively applied in support of policy making in the field of air pollution control.

[34] Holland et al. (2002) assume linear dose response relations for crops, but this linearity has not been confirmed for ecosystem responses to ozone.

4.4 Uncertainties in the economic analysis of ecosystem changes

Applying ecosystem approaches in support of policy formulation requires analyses of the economic costs or benefits of a change in ecosystem services supply due to changes in air pollution. Much attention has been devoted in recent decades to the development of methodologies for the valuation of ecosystem services, in particular those services not traded in the market (e.g. Daily et al., 2009).

In spite of these advances, there are still important uncertainties remaining in the field of ecosystem services valuation. Vatn (2005) describes the following four main points of general concern regarding the valuation of ecosystem services (i) a lack of full information on ecosystem services; (ii) value incommensurability; (iii) the problem of composition; and (iv) the income-dependency of willingness to pay estimates. A brief overview of these points is provided below.

(i) A lack of full information on ecosystem services. A lack of information is a frequent constraint to ecosystem valuation. For instance, there may be only approximate indication of the actual use level of the service, its marginal value in case of strong changes in supply, ecosystem dynamics and how they influence future supply of the service, etc. These constraints progressively increase at coarser scales and with increasing complexity of the ecosystem. In particular, at the European scale there may be a lack of data on the response of specific ecosystem types to pollution levels, and there may be unknown price effects related to changes in ecosystem services supply (for instance changes in crop damages).

(ii) Value incommensurability. Value incommensurability means that different types of values, for instance the values related to biodiversity, cultural functions of ecosystems and values derived from products harvested in an ecosystem, cannot be measured on one and the same scale. This argument is based on the observation that individuals have different motives for managing ecosystems, and that they therefore have difficulty in interpreting services and values along one dimension – as in the case of comparing positive effects on biodiversity with negative effects on timber production.

(iii) The problem of composition. The problem of composition indicates that the supply of an ecosystem service is always dependent on the functioning of the ecosystem supplying the service, and that demarcating parts of the environment for the purpose of valuation may lead to underestimation of the value of the ecosystem at large.

(iv) The income-dependency of willingness-to-pay estimates. The income-dependency of willingness-to-pay (WTP) estimates is a concern where there are large income discrepancies between different stakeholders. The WTP estimate is bound by the income of the respondent and restricts the articulation of unrealistically high WTP statements in a contingent valuation study (Arrow et al., 1993). Nevertheless, the wide range of WTP estimates for avoiding health impacts, as also indicated by the differences between median and average values in this regard, indicates the potential magnitude of the uncertainties involved. It can be expected that uncertainties will be comparably high for WTP estimates related to ecosystem impacts (e.g. on species richness).

There is increasing experience with the valuation of some ecosystem services, in particular provisioning services and several of the regulating services such as water regulation and carbon sequestration as well as recreation and tourism. Increasingly, the outcomes of such studies are used in the policy formulation process, including in the EU (TEEB, 2010). Nevertheless, there are still important methodological challenges remaining to the valuation of a whole range of ecosystem services. For the services affected by air pollution in Europe, these challenges are particularly relevant with regards to the impacts on biodiversity, and it is questionable if impacts of air pollution on this service can be meaningfully valued in economic terms. For carbon sequestration, there is a factor 2 to 3 uncertainty on the appropriate price to use for a unit of carbon, on the basis of a comparison of the different estimates for the marginal social damage costs of carbon, the prices for which carbon is traded in the carbon market, and the prices included in governmental guidelines for Environmental CBA. For recreation, there are currently insufficient data to establish the general relation between a decline in the recreational quality and the loss of economic value in particular for forests. For lakes, several case studies have been done on willingness to pay to avoid a reduction in water quality, but insufficient to establish a relation that is applicable at a wider, let alone European, scale. Hence, further advances in valuation methodologies are required before European wide economic analyses of the effects of changes in air pollution levels can be conducted.

5 - Policy and Scientific Implications

There are currently still relatively few studies that quantify impacts of air pollution on ecosystem services supply in monetary terms. For example, Holland et al. (2002) analyse impacts of ozone on crop production, CIAM (2007) includes work on the impacts of ozone on timber production, and Smart et al. (2011) analyse the costs of ammonia leading to the release of greenhouse gas in the UK. Based on a review of the current understanding of ecosystem services modelling and valuation in the context of air pollution, CONCAWE recommends the following:

1 - There is a need to better understand ecosystem effects before they are included in CBAs of European air policy options. Positive effects of air pollution on ecosystem services need to be accounted for, for instance nitrogen deposition will, in N limited ecosystems, generally lead to enhanced supply of ecosystem services such as timber production and carbon sequestration.
2 - There is a need to examine how marginal costs and benefits of changes in ecosystem services supply resulting from changes in air pollution can be analysed. An important question is what effect passing critical loads thresholds will have on ecosystem functioning and subsequently on the supply of ecosystem services. This effect is likely to differ for different ecosystem types and different types of ecosystem services.
3 - The effects of complex ecosystem dynamics need to be better understood prior to including ecosystem impacts on CBA models. In particular, there are likely to be complexities resulting from interactions between the multiple stressors on ecosystems including air pollution. In addition, there may be lag effects occurring, for example reducing air pollution deposition rates to below critical load levels may not immediately lead to restoration of ecosystem functioning or ecosystem services supply. These lag effects will affect the cost-benefit ratio of different policy options.
4 - There is a need to better understand society's willingness to pay for biodiversity. Reducing eutrophication, in particular, may lead to lower timber production and carbon sequestration in nitrogen limited forest ecosystems, but may enhance biodiversity in these forests. A question is how biodiversity effects and negative impacts on other services can be compared.
5 - Given that the uncertainties involved are very significant, as outlined in this Appendix, there is a particular need to conduct thorough uncertainty and sensitivity analyses to indicate the robustness of the assessments before they are used in support of policy making.

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