European Commission

Research Findings in support of the EU

Air Quality

Review

Research and Innovation

EUROPEAN COMMISSION

Directorate-General for Research and Innovation
Directorate I— Environment
Unit I.4 — Climate Change & Natural Hazards

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ACCENT *Plus*
ATMOSPHERIC COMPOSITION CHANGE
THE EUROPEAN NETWORK

European Commission

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Air Quality

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Cataloguing data can be found at the end of this publication.

Luxembourg: Publications Office of the European Union, 2013

ISBN 978-92-79-29457-0

doi 10.2777/98974

Foreword

Significant progress has been made in Europe during the last decades to improve air quality. However, big challenges still lie ahead if we are to reach a position where we can guarantee a sufficient level of protection against air pollution. According to the 'Attitudes of Europeans towards air quality' Eurobarometer, air quality remains one of the main concerns of European citizens.

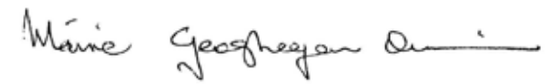
Particulate matter, ozone and nitrogen oxides are, today, the most important air pollutants in Europe, posing serious risks to human health and the environment, in particular for their impact on climate change. According to recently published studies[1], 430,000 premature deaths can be related to particulate matter exposure.

Research is at the very heart of the process launched by the European Commission to undertake a full revision of its air quality policy. In 2013, this review aims to produce a robust EU Clean Air package, updating existing policies and Directives in accordance with the latest science.

In support of this review, the Directorate General for Research and Innovation launched an intense consultation with some of the main EU-funded research projects in this area. The findings are set out in this present report, which contributes to strengthening the essential two-way dialogue between science and the policy-making process.

Through Horizon 2020 – the future framework programme for research and innovation for the period 2014-2020 – the European Commission will tackle the main policy priorities of the Europe 2020 Strategy for Smart, Sustainable and Inclusive Growth, covering the major concerns shared by citizens in Europe and beyond. Research in this field and the development of technological options and strategies to fight against air pollution and climate change represents a part of Horizon 2020.

I would like to express my gratitude to the many scientists who have contributed to this publication.



Máire Geoghegan-Quinn
Commissioner for Research, Innovation and Science

1 Global disease burden 2010 (The Lancet, December 2012)

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Executive summary

Introduction

This report provides a summary of the policy relevant messages from the research and monitoring communities as an input to the EU Air policy implementation and review process. The approach in distilling the messages has been to provide a minimum of background science as context, explain the message simply and identify what remains unclear or not understood to show the levels of confidence of further policy actions. Lastly we identify further research needs to address these uncertainties.

The aim of the EU Air policy implementation and review process is to provide:

"a robust EU Clean Air package, updating existing policies and directives including the National Emission Ceilings Directive according to latest science, and outlining further cost-effective measures to move much closer to the related 6EAP's objective: *to achieve levels of air quality that do not result in unacceptable impacts on, and risks to, human health and the environment."*

This process therefore requires an understanding of the messages from research and monitoring activities throughout Europe and developments in the wider global science community. This review, focuses on research and assessment activities within Europe and has the following two objectives:

To identify key scientific messages relevant for the revision and implementation of EU Air Quality legislation

To identify limitations to knowledge and emerging issues

The review has been coordinated by the European Commission Directorate General for Research & Innovation with contributions from the JRC, DG ENV, DG SANCO, the European Environmental Agency (EEA), the World Health Organisation (WHO) and the International Institute for Applied Systems Analysis (IIASA). The hearth of the report has been produced by several European research projects (listed in the previous section).

The wider global literature has been used to provide a wider perspective and context for the review.

Air Quality legislation

Air Quality directives, the National emissions ceilings directives and the UNECE Gothenburg protocol provide the main control instruments.

The science review process

The report has been elaborated through a very intensive consultation process that took a little over a year from Start Oct 2011-to complete Jan 2013. The preliminary results have been presented to the group of stakeholders chaired by DG ENVIRONMENT.

The chapters are recognisably the product of scientists, the messages are not policy prescriptive and do not necessarily take into account the complexity of the political process involved in the revision process, this is the nature of science.

The policy messages are broad, and consider effects on human health, ecosystems and climate.

The format of this Executive summary is provided by a set of questions and brief answers derived from the review. Each of the questions and further background context are addressed in more depth in the Chapters following the summary.

Air Quality and Health (Bert Brunekreef at al.)

Q1. Do we know the components of PM responsible for health effects and the relative importance of natural and anthropogenic components?

So far, it has not been possible to identify one or two components, which are primarily responsible for the harmful effects of PM.

It has not been possible either to conclusively show that specific PM components, at relevant outdoor concentrations, are harmless. Notwithstanding, the evidence is strong for reducing primary combustion particles.

There is increasing evidence that 'natural' components such as windblown dust are harmful to health. The PM mass concentration varies, e.g. with distance to major source regions such as the Sahara.

Q2. What is the relative importance of background ozone and peak ozone for health effects?

Epidemiological studies find associations between ozone and mortality down to very low concentrations in time series studies. Lowering background concentrations is therefore as important as reducing peak exposures.

Q3. Is a single health-related objective covering all pollutants scientifically credible? And could it be used to drive policy?

Air Quality Indices do exist, but mostly, such indices are used as public information tool, and not as tools to manage air quality, or to study effects of combined air pollution exposures

Q4. Are the relative contributions to health effects from indoor and outdoor exposure correctly coupled to the sources of the pollutants responsible?

Fine particles from outdoor origin are the largest single contributor to the overall burden of disease from indoor air pollutants in Europe.

Particulate Matter (S.Fuzzi et al.)

Q1. The size of particulate matter (PM), PM10, PM2.5 or PM1. What matters and which PM size fraction should be the focus of new regulatory effort?

Both fine and coarse particles show a link to health effects, although on different time scale exposure: fine particles effects are seen for both long and short term exposure, while coarse particles link only to short term exposure.

The metrics for fine and coarse particles, according to the present legislation, are PM2.5 and PM2.5-10, respectively. Potentially, PM1 could be more effective in separating fine and coarse aerosols based on chemical composition and sources, although there will always be overlap between accumulation-mode and coarse-mode particles in the 1 – 3 μm size range.

Q2. Composition, do we know the link between aerosol composition and health effects?

The air concentration of PM mass is the parameter that has been most closely linked to health effects.

There is suggestive evidence of a BC concentration link to short and long term health effects. It would be useful to include it on the list of monitored/regulated pollutants.

A few classes of compounds in PM, representing a minor fraction of total mass, are known to induce relevant health effects: transition metals and aromatic organic compounds (PAHs). These compounds should be subject to ad-hoc regulation with time and space specific objectives.

Primary biogeny aerosol particles (PBAPs) are associated with negative health impacts. Although they have a natural origin, their emission and transport can be affected, directly and indirectly, by anthropogenic activities.

Recent studies show relevant health effects of primary and secondary organic aerosol, especially of the more volatile fraction.

Q3. What are the most important sources to control and in what order?

There is insufficient evidence to set ambient limits for all the potentially toxic components of PM. There is evidence to suggest traffic sources of primary PM are important. Although measures on total PM mass will enable existing limit values to be met, this may not be an optimal strategy to improve public health, and consideration should be given to an ambient limit for primary traffic PM components.

Reduction of PM levels could be achieved by reduction of secondary inorganic aerosol through reduction of their precursors (ammonia, VOCs, nitrogen oxides, and sulphur dioxide). In particular, NH_3 emissions have been subject to very little control to date, including the Gothenburg revision. Control methods are available for NH_3 and represent a cost effective measure to reduce PM levels.

Suggestive evidence, not definitive at this stage, points towards traffic emissions as potentially more potent than other sources for their health impact. An efficient surrogate of traffic-related emissions is Black Carbon.

Non-exhaust traffic emission is not regulated and is gaining special relevance, especially when exhaust emissions have been abated as in recent years. This source of emissions is a significant source of transition metals.

Q4. Hot spots or the background, which should be the priority for control of PM sources and precursors?

The greatest benefit of further control measures would be in the reduction of background PM concentrations: regional air masses affect urban areas where large fraction of population is exposed, exceedance episodes are more frequent where high background PM levels are observed.

Specific emission reductions for hot spots (e.g. increasing combustion plant efficiency, introducing district heating network, energy-efficiency modernization) are indicated, especially where they differ greatly from regional background concentrations.

Reduction of long-range transport of pollutants requires the control of emission sources at local, regional, and continental scale. Regional and international cooperation is necessary to implement successful emission control strategies.

Emission inventories are not homogenous on temporal and spatial scales, and their consistency across scales needs to be improved.

Q5. Can the control of PM sources be regulated to maximise the benefits for health and climate effects?

There are control measures of PM sources that can simultaneously benefit air quality, health, and climate change, these include measures that reduce energy consumption and promote the use of non-combustion renewable energy.

A series of PM control measures have been identified to maximize the benefits for health and climate effects: promotion of low emission vehicles, increase of energy efficiency, use of particle filters for diesel vehicles, reduction of shipping emissions, and enforcement of existing ban of open field burning.

Most of these measures focus on reduction of black carbon emission sources, due to its concomitant negative impact on health and climate.

Ozone (P.S. Monks et al.)

Q1. Can the background contribution to ecosystem and human health effects be quantified?

The contribution of background ozone to stomatal flux or SOMO35 can be quantified but model uncertainty is large, partly because of uncertainties attributed to externalities such as precursor emission or long range transport in addition to the uncertainties of the photochemical models.

Background ozone makes a relatively small contribution to the measures of peak exposure (AOT40 and SOMO35) but it dominates the mean ozone concentration and contributes significantly to its trend in Western Europe.

New proxies for ozone exposure should be considered as peak ozone may not be an effective measure of impacts on human health and the environment, especially seasonally.

Q2. What is the contribution of background ozone to effects on human health and ecosystems?

If ozone precursor emissions were uniformly reduced globally by 20%, over half of the mortalities avoided due to ozone in Europe could be attributed to non-European emissions

Transboundary emission reduction of ozone precursors can significantly reduce downwind crop production loss. For the case of Europe, reductions in anthropogenic NOx emissions from N. America give the largest effect in crop production loss.

Q3. What is the relationship between control of NOx and VOC sources in Europe and changes in ozone exposure and effects?

As NOx emissions are reduced, ozone generally increases within the NOx-saturated parts of Northern Europe. It may be more relevant to discuss total oxidant in this context.

Megacity based abatement strategies will not substantially impact regional ozone; there is a need for more than a city-based ozone precursor reduction policy.

Q4. How much benefit would control of methane deliver in mitigating ozone effects in Europe?

Methane reduction has potential win-win benefits for air quality and climate

Substantial benefits for ecosystem and human health would result from large reductions in global methane, but the effect of controls in Europe alone would be relatively small.

Q5. Does ozone remain a problem in Europe and what is the most effective geographical scale for further control measures?

Ozone concentrations across Europe remain at levels that are detrimental to human ecosystem health.

External, non-European sources and CH_4 exert a larger effect than EU ozone precursor emissions on annual mean EU O_3, thus control of European O_3 precursors is not a very effective control measure in the absence of hemispheric controls

In urban areas, the coupling of NO_2 and O_3 need to be treated as total oxidant for control purposes.

There are clear and quantifiable co-benefits of treating air quality and climate together.

More attention needs to be given to treating ozone as a hemispheric transboundary issue.

Nitrogen (M.A. Sutton et al.)

Q1. How do the effects of nitrogen emissions on climate change interact with air quality?

Nitrogen pollution effects on climate include warming from N_2O and the N contribution to tropospheric ozone, and cooling from the N effect on biosphere CO_2 exchange and from N_r containing aerosol.

Overall, a net cooling effect is estimated for present emissions, though the warming effect of N_2O is the longest term consequence, while the cooling effect of CO_2 uptake may tend to saturate over future decades.

Improving Nitrogen Use Efficiency (NUE) must be a central element in strategies to reduce emissions of N_2O, NH_3 and NO_x simultaneously, demonstrating a clear synergy between air quality and climate policies. Better N management for air quality will therefore simultaneously deliver reductions in N_2O emissions.

Q2. Could ecosystem effects of NH_3 and the potential of a new AQ limit value be used in delivering Habitats Directive commitments?

The use of a revised AQ limit value for NH_3 could provide substantial benefits for the habitats directive.

Further reductions in N_r emissions and NH_3 in particular would reduce effects of N_r on the Nature 2000 network.

The critical level for ammonia has been revised and has been adopted by the UNECE CLRTAP. The critical level could provide the basis to define an NH_3 limit values applicable over the territory of the European network of Special Areas of Conservation (SACs).

An NH_3 limit would provide benefits for habitats, climate and health.

Q3. What is the overall economic cost of nitrogen in the EU environment?

The overall costs are estimated at €70–€320 billion per year, of which 57 to 59% is related to air pollution effects on human health. The total damage of all reactive nitrogen losses to the environment cost equates to €150–€750 per person.

Revision of ammonia abatement costs across the EU (mid scenario for preparation of the Gothenburg protocol) indicates a cost of €0.6 billion per year, equivalent to 0.8 Euro/kg N abated, highlighting that agriculture is a cost-effect sector for reducing Nr emissions to the environment.

Effects of AQ on ecosystems are of comparable order of magnitude to human health effects.

Q4. How can we improve the valuation of air pollution threats to ecosystems: demonstrating the dose-response-valuation chain?

The concept of Ecosystem Services (ES) and research into the quantification of specific services has the potential to close the existing gap of established valuation approaches for ecosystem effects from air pollution.

Although a first valuation of ecosystem effects has been made in the European Nitrogen Assessment, the coupling of dose-response relationships to damage valuation is not yet sufficiently developed for inclusion within Integrated Assessment Models, for example to provide a basis to optimise control measures.

Q5. What are the relative costs and benefits of controlling NO_x and NH_3 emissions?

With reducing NO_x emissions over the last decades, NH_3 emissions are now a more powerful driver of effects on ecosystems than oxidized nitrogen, making future control of NH_3 emissions a priority.

As many measures for control on NO_x emissions have already been implemented, further technical measures become increasingly expensive. By contrast, at the European scale, only a few of the available technical measures for NH_3 have so far been implemented, with many low-cost measures still available. Ammonia therefore offers substantial 'low-hanging fruit' for future air pollution controls.

Estimated benefit-cost ratios for technical measures (as included in the GAINS model) justify further European reductions in emissions of around 100-400 kt N for NO_x and around 800-1100 kt N for NH_3.

Air Quality and Climate (G. Brasseur et al.)

Q1. What are the synergies between air quality and climate change?

In many countries, the mitigation policies for climate and air quality are quite separate and ignore therefore the relations between them.

Major greenhouse gases originate from the same sources as air pollutants, and a coordinated abatement strategy could provide an effective way of securing benefits for both policy areas.

Q2. Can the climate change and air quality antagonism and synergies be quantified?

Great potential exists for the application of air pollution mitigation strategies to contribute to climate policies.

Removing black carbon and reducing methane would simultaneously benefit air quality and climate in the short term.

IPCC RCP assumptions effectively return aerosols to pre-industrial levels by 2100 in all scenarios, greatly enhancing climate warming; however, embedded in the RCP emission scenarios is the assumption that stringent air quality policies will be adopted. This important assumption will not necessarily happen.

Q3. What is the effect of climate policy scenarios on achievement of air quality objectives?

Reduction in the emissions of greenhouse gases, in particular methane, and of absorbing aerosols, especially black carbon, should contribute to an improvement of air quality, specifically to a reduction in the ozone concentration.

Reduction of SO_2 emissions will increase global warming.

Q4. Are there regions particularly sensitive to European pollutant emissions?

Yes, the Arctic:

Transport of pollutants from Europe to the Arctic, for example, is relatively efficient especially in the lower troposphere, and European emissions therefore have a strong impact on the near-surface concentrations of short-lived climate forcers in the Arctic

Integrated assessment (R. Friedrich et al.)

Q1. How can policies be developed to regulate impacts of both air pollution as climate change simultaneously?

Integrated assessment of air pollution and climate policy is necessary to maximize the benefits of investment in control measures.

To assess biodiversity losses or gains in monetary terms, some approaches (e.g. in the context of ecosystem services) have been developed that should be tested for their usefulness in the review process.

To be able to compare climate change with air pollution impacts, a conversion into monetary indicators is required to consistently quantify costs and benefits of EU climate policy aims.

Q2. Are the scales of assessment for the different pollutants (from local scales of urban air quality to the global scale assessment of radiative forcing by GHG) properly considered in IA?

Methods for a better spatial and temporal resolution of emission data have been developed and could be used to improve the quality of spatially and temporally resolved emission data.

Q3. Should the assessment of non-technical measures be included in an integrated assessment?

As the potential of technical measures, i.e. measures that reduce emission factors, is limited, an assessment of measures should include non-technical measures, i.e. measures that change the behaviour and thus emissions.

Introduction

This summary document has been written to highlight research findings on a series of air quality issues in Europe. The list is not exhaustive and there is a focus on particulate matter, ozone and nitrogen in which earlier air quality legislation has led to substantial reductions in emissions of their precursor emissions. However, there remain substantial issues for each of these pollutants and further measures are required to reduce effects on human health and ecosystems. In addition, interactions between air quality and climate are included as each influences the other and policies developed without recognising these interactions miss important opportunities to maximise the environmental benefit of a wider view. Lastly, Integrated Assessment has been a key tool in providing strategies for discussion in the development of new legislation and is included in this review.

Air Quality and human health has been the main driver of recent air quality policy development and as it is an overarching issue, it is presented first.
The objectives of the document are to highlight research findings that are relevant to policy development, recognising that the research teams still do not have all the answers and the findings do not always provide easy policy decisions. A short list of priorities for further research is therefore included at the end of the document to highlight timely issues to address with additional research.

The research findings outlined in this document are restricted to those considered helpful in the current policy review process. The findings are structured by issue and also by pollutant as no single taxonomy entirely separates the pollutants and issues.

The objective of the summary document is to communicate the key policy relevant outputs of the research projects involved in a clear and straightforward style, supported by links to peer reviewed publications to provide the evidence in support of conclusions made. Detailed descriptions of the research will be referenced rather than included explicitly in the document. In this process it is necessary to state what is known and what remains unknown. Thus, where current knowledge is not sufficient to provide a clear answer, the uncertainty is highlighted, and where possible, specific additional research requirements are identified.

Chapter 1
Air Quality and Health

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Introduction

Outdoor air pollution remains a threat to human health in Europe. The pollutants of most concern are particulate matter, ozone and nitrogen dioxide. Existing EU limit values and WHO Air Quality Guidelines for these pollutants are regularly exceeded, and a significant health burden has been estimated to still exist. Their impact undermines social and economic development at the community, national and global levels. Particles less than 2.5 microns (PM2.5) are considered most hazardous from a health point of view, but larger particles (PM10) also cause harm and are subject to EU regulations. The harmful effect of particulate matter is caused partly by local and systemic inflammation in response to the particles themselves and partly from substances on the particles which may be absorbed via the lungs. Short term health effects include increases in daily mortality and hospital admissions and worsening of asthma, while longer term effects include loss of life expectancy through cancer, cardiovascular and chronic respiratory diseases. Vulnerable groups such as children and older people are more susceptible to poor air quality.

This chapter focuses primarily on contributions from recent EU funded studies and is closely coordinated with WHO to avoid overlap, gaps or inconsistencies.

Q1. Do we know the components of PM responsible for health effects and the relative importance of natural and anthropogenic components

Many studies have addressed this broad question over the last 15 years. So far, it has not been possible to identify one or two components which are primarily responsible for the harmful effects of PM. It has not been possible either to conclusively show that specific PM components, at relevant outdoor concentrations, are harmless. Notwithstanding, the evidence is strong for reducing primary combustion particles. Finally, natural components of PM such as observed during Sahara dust events have recently been associated with a range of adverse health effects.

The ESCAPE and TRANSPHORM studies are specifically addressing the issue of PM components and sources responsible for health effects. In the context of a series of on-going population cohort studies, these FP7 projects have generated very detailed exposure matrices allowing the investigation of the relative importance of different PM sizes and components. Specifically, the focus is on PM2.5, PM10, coarse PM, Black Carbon, PM elemental composition, PM PAHs, and PM oxidative potential.

Results of the measurements of PM10, PM2.5, coarse PM, Black Carbon and NO_2 have been published (Eeftens et al. 2012a; Cyrys et al. 2012), see Figures 1.1 and 1.2. They make clear that especially the US EPA NAAQS for PM2.5, and the EU Limit Value for NO_2 are still massively exceeded in many areas in Europe. Also, a paper has been published showing that in most of the study areas, Land Use Regression models were able to explain a large proportion of the spatial variation of the annual average concentrations of the respective PM metrics (Eeftens et al. 2012b).

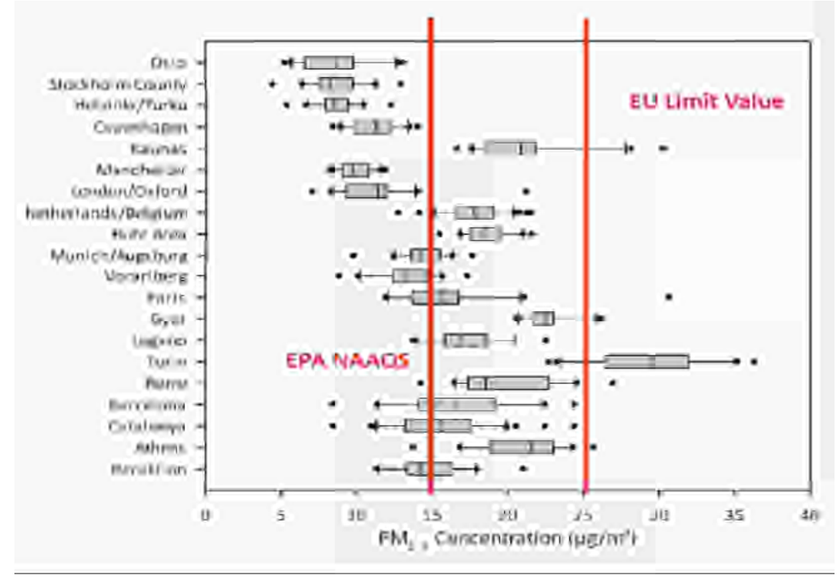

Figure 1.1 Annual mean PM2.5 concentrations measured in the ESCAPE project (Eeftens et al. 2012a).

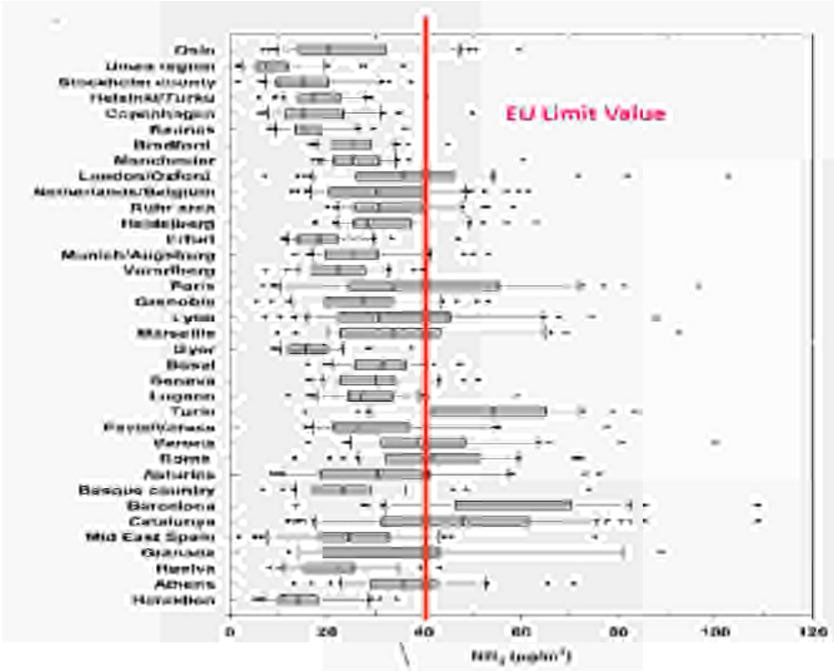

Figure 1.2 Annual mean NO_2 concentrations measured in the ESCAPE project (Cyrys et al. 2012).

The health analyses of the ESCAPE project are nearing completion, but publication of results in the peer-reviewed literature is not yet completed.
In this report, as an example, we show four tables that are part of the ESCAPE publishable summary as submitted to DG Research and Innovation in January 2013. Table 1.1 shows that across the participating cohorts, Natural Cause Mortality was significantly associated with PM10 and PM2.5. The effect estimate for PM2.5 was a 7% increase for each 5 µg/m³ increase in PM2.5.

Table 1.1 Association between natural cause mortality and air pollutants, jointly estimated from 19 (PM) and 22 (NO_2) European cohort studies

Pollutant	Relative Risk (95% C.I.)
NO_2 full range (per 10 µg/m³)	1.01 (0.99-1.03)
NO_2 below 40 µg/ m³	1.02 (0.99-1.05)
NO_2 below 30 µg/ m³	1.02 (0.99-1.05)
NO_2 below 20 µg/ m³	1.01 (0.91-1.11)
PM2.5 full range (per 5 µg/ m³)	1.07 (1.02-1.13)
PM2.5 below 25 µg/ m³	1.06 (1.00-1.12)
PM2.5 below 20 µg/ m³	1.07 (1.01-1.13)
PM2.5 below 15 µg/ m³	1.04 (0.98-1.11)

Table 1.2 shows what happens to the effect estimates for PM2.5 and NO_2 when study participants having exposure estimates higher than some specified threshold were excluded from the analyses. The results show that effect estimates for NO_2 remain elevated and (borderline) significant even when all subjects with concentrations higher than 30 µg/m³ are excluded. The current annual mean Limit Value is 40 µg/m³. For PM2.5, effect estimates remain significantly elevated when all subjects with concentrations higher than 20 µg/m³ are excluded. The current Limit Value is 25 µg/m³. So especially for PM2.5, significant associations with Natural Cause Mortality were seen over concentration ranges which were entirely well below the current Limit Values.

Table 1.2 Association between natural cause mortality and NO_2 and PM2.5, excluding subjects above successively lower thresholds of exposure.

Pollutant	Relative Risk (95% C.I.)
PM10 (per 10 µg/m³)	1.12 (1.01-1.25)
PM2.5 (per 5 µg/m³)	1.13 (0.98-1.30)
PM absorbance (per m^{-1})	1.10 (0.98-1.24)
NO2 (per 10 µg/m³)	1.03 (0.97-1.08)

Table 1.3 shows a similar analysis for associations found with Coronary Events. Significant associations across cohorts were found for PM10, whereas associations with PM2.5 and PM absorbance were elevated without reaching statistical significance. Table 1.4 shows a threshold analysis for Coronary Events. Even after excluding all study participants with PM10 concentrations higher than 20 µg/m³, or PM2.5 concentrations higher than 15 µg/m³, significant associations with Coronary Events remain, again showing that these associations persist over concentration ranges which are entirely well below current annual mean Limit Values for PM10 and PM2.5. Most relevant are the findings from the Swiss SAPALDIA study confirming that the reduction in PM10 – which are highly correlated with PM2.5 in Switzerland – that occurred during the cohort follow-up (1990 to 2001 due to clean air policies was paralleled by improvements in various indicators of health. These improvements were all seen at levels of PM10 well below the current EU limit value of 40 µg/m³ (annual mean) (Kuenzli et al. 2009; Schindler et al. 2009).

Table 1.3 Association between incidence of coronary events and air pollutants, jointly estimated from 11 European cohort studies

Pollutant	Relative Risk (95% C.I.)
PM10 full range (per 10 µg/m³)	1.12 (1.01-1.25)
PM10 below 40 µg/m³	1.15 (1.02-1.30)
PM10 below 30 µg/m³	1.12 (0.98-1.27)
PM10 below 20 µg/m³	1.20 (1.01-1.41)
PM2.5 full range (per 5 µg/m³)	1.13 (0.98-1.30)

Secondary inorganic aerosols (predominantly nitrates and sulphates) make up a sizable fraction of PM2.5 in many areas. Epidemiological studies have frequently found these to be associated with adverse health effects. This is in contrast to findings from laboratory studies where exposure to higher than ambient concentrations of sulphates and nitrates generally fail to elicit much of a biological effect. A recent review provides a number of potential pathways through which sulphates and nitrates may produce the biological effects observed in epidemiological studies (Kelly and Fussel 2012).
Black carbon (BC) has received special attention, as it is a more appropriate metric to evaluate health risks of traffic related air pollution than PM mass. A recent review (Janssen et al. 2011) has summarized the evidence and provides considerations for establishing health based guidelines and standards for BC.

Table 1.4 Association between incidence of coronary events and PM10 and PM2.5, excluding subjects above successively lower thresholds of exposure.

Pollutant	Relative Risk (95% C.I.)
PM2.5 below 25 µg/m^3	1.23 (1.04-1.46)
PM2.5 below 20 µg/m^3	1.23 (1.04-1.46)
PM2.5 below 15 µg/m^3	1.19 (1.00-1.42)

Other studies have evaluated the role of transition metals such as Vanadium and Nickel, the role of organic carbon, polycyclic aromatic hydrocarbons etc. Such studies typically evaluate the associations between some PM component concentration and some health endpoint directly or indirectly by examining whether the association between PM mass and health is different in areas where or at times when PM composition is different.

There is increasing evidence that 'natural' components such as windblown dust are harmful to health (Mallone et al. 2011). The relative contribution of natural components to the PM mass concentration varies, e.g. with distance to major source regions such as the Sahara.

The most recent assessment of the US EPA has summarized the state of the art in a 2010 document as follows:

"In considering whether the currently available evidence provides support for retaining, revising, or supplementing the current PM2.5 mass-based indicator, we first conclude that it is appropriate to consider retaining PM2.5 as the indicator for fine particles. Secondly, we conclude that the currently available evidence does not provide a sufficient basis for supplementing the mass-based PM2.5 indicator by considering a separate indicator for ultrafine particles. We also conclude that the currently available evidence is too limited to provide support for considering a separate indicator for a specific PM2.5 component or group of components associated with any source categories of fine particles or for eliminating any individual component or group of components associated with any source categories from the mix of fine particles included in the PM2.5 mass-based indicator."(EPA 2011).

This assessment by EPA was finalized before the evidence on health effects associated with measures of Black Carbon was reviewed (Janssen et al. 2011), and it seems prudent that, in addition to maintaining Limit Values for fine particle mass, serious consideration is given to this metric in standard setting and policy.

A recent literature review and time series analysis provides guidance for design and analysis of future studies to resolve the role of PM components better (Levy et al. 2012).

The issue of which are the most important sources of PM to control too has received much attention over the past 1-2 decades. There is consensus that primary combustion particles need to be controlled – and much has been achieved already reducing primary PM emissions from power plants, industrial facilities and motor vehicles.

Despite reductions in PM emissions from motor vehicles, studies continue to find links between traffic related air pollution and a range of adverse effects including premature mortality, heart disease, COPD and asthma. Possibly, ultrafine particles play an important role here as these are biologically very active and reach high concentrations near busy roads. As shown in APHEKOM (Perez et al. 2013), the health burden of living close to busy roads is very substantial in European cities due to the very high fraction of people living under such conditions. The APHEKOM project has applied innovative health impact assessment methods to take into account the long-term impact on the development of chronic diseases from living near busy roads. It also evaluated the monetary costs associated with this impact. The project determined that living near these roads could be responsible for some 15-30 per cent of all new cases of asthma in children, and of COPD and CHD in adults 65 years of age and older. The report also points out that the associated economic burden could total €300 million every year. The Swiss SAPALDIA study showed that the decrease in traffic-related near-road PM resulted in lower rates of adult-onset asthma. It is difficult, however, to separate the role of ultrafines from the role of other near-road traffic-related pollutants. As vehicle fleets in the real world are always a mix of old & new, it is very difficult to say whether the newest emission controls such as diesel particle traps are sufficient to eliminate adverse health effects. Experimental work is showing encouraging results, however (Lucking et al. 2011).

A systematic review of studies on combinations of components and source apportioned PM has not found a single or a small number of major sources to be responsible for most of the PM health effects (Stanek et al, 2011). Rather,

different components and sources were found to be most closely related to a range of adverse health effects in different studies covered in this review, suggesting that controlling PM mass may still be the best option for reducing health effects attributed to various characteristics and components of PM.

Summary

> So far, it has not been possible to identify one or two components, which are primarily responsible for the harmful effects of PM.

> It has not been possible either to conclusively show that specific PM components, at relevant outdoor concentrations, are harmless.

> Notwithstanding, the evidence is strong for reducing primary combustion particles.

> There is increasing evidence that 'natural' components such as windblown dust are harmful to health. The PM mass concentration varies, e.g. with distance to major source regions such as the Sahara.

Q2. What is the relative importance of background ozone and peak ozone for health effects?

The adverse effects of peak ozone concentrations on the respiratory system have been well understood for several decades. More recently, short-term variations in ozone have also been shown to be associated with short-term changes in mortality (Bell et al. 2005). In comparison, much less is known about effects of long-term exposure to elevated background ozone concentrations on human health. Some studies have found effects on the development of asthma, and one major study has shown a relationship with respiratory but not cardiovascular mortality (Jarrett et al. 2009). No European studies have looked at long-term effects of elevated background O_3 concentrations recently.

Summary

> Epidemiological studies find associations between ozone and mortality down to very low concentrations in time series studies.

> Lowering background concentrations is therefore as important as reducing peak exposures.

Q3. Is a single health-related objective covering all pollutants scientifically credible? And could it be used to drive policy?

This question is about developing one single indicator that could be used efficiently to drive air pollution abatement policies. The FP7 PASODOBLE project has recently developed an Aggregate Risk Index for short term effects of major air pollutants on mortality and morbidity (Sicard et al. 2012). It takes the simple form of

$ARI = \Sigma_i (RR_i - 1)$

where RR_i is the relative risk per unit concentration change in pollutant i. Arbitrarily, the risk is categorized as 'low' when the increase is < 12%; 'moderate' at an increase of ≥12% and < 21%, 'high' at risk increases of 21-29.9% and 'very high' at risk increases of ≥30%. The relative risk numbers were derived from recent meta analyses as published by WHO for PM2.5, PM10, SO_2, O_3 and NO_2.

Some countries or cities outside Europe (such as Hong Kong) have used an Air Pollution Index to summarize the observed concentrations of a number of major air pollutants into a single number. The Hong Kong Index is based on respirable particles, ozone, carbon monoxide, sulphur dioxide, nitrogen dioxide and lead. The Canadian Air Quality and Health Index is based on PM2.5, NO_2 and ozone. Mostly, such indices are used as public information tool, and not as tools to manage air quality, or to study effects of combined air pollution exposures. As the available indices are based on risk estimates for single pollutants, future studies should include aggregate indices of exposure to shed more light on the usefulness of a single health related objective.

The APHEKOM project has made detailed analyses of the world and emerging European literature on the health effects of air pollution, and found no evidence to support that major urban air pollutants in Europe have more or less of an effect on public health than the same pollutants studied elsewhere (Perez et al. 2013). The project has also made a quantitative analysis of the gains in life expectancy that could be achieved in different European cities if they were to comply with the WHO Air Quality Guideline for PM2.5. Figure 1.3 shows that in many cities which are still highly polluted, such gains could be substantial. For details, www.aphekom.org

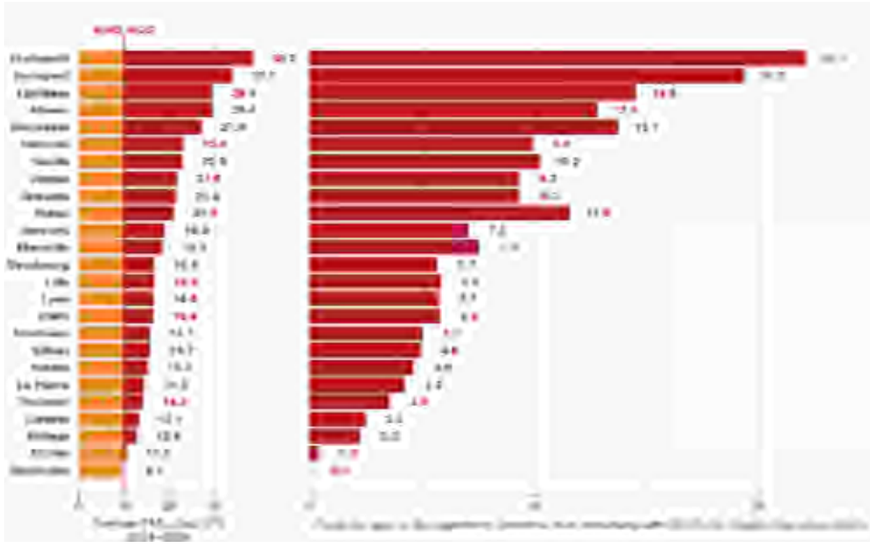

Figure 1.3 Predicted average gain in life expectancy (months) at age 30 in 25 Aphekom cities for a decrease in average annual level of $PM_{2.5}$ to 10µg/m³ (WHO's Air Quality Guideline). Source: www.APHEKOM.org

Summary

> Air Quality Indices do exist, but mostly, such indices are used as public information tool, and not as tools to manage air quality, or to study effects of combined air pollution exposures

Q4. Are the relative contributions to health effects from indoor and outdoor exposure correctly coupled to the sources of the pollutants responsible?

It remains important to realize that most people spend over 90% of their time indoors, and that exposure to pollutants of outdoor origin takes place indoors primarily. Exposure may be modulated by ventilation and filtration so that the same outdoor concentration may have a different effect in communities with, e.g. much air conditioning compared to communities without (Bell et al. 2009). The IAIAQ study estimated that fine particles from outdoor origin are the largest single contributor to the overall burden of disease from indoor air pollutants (http://ec.europa.eu/health/healthy_environments/docs/env_iaiaq.pdf). Indoors, exposure to air pollutants varies with presence or absence of sources such as building material, heating, household products, cleaning agents, and with quality of housing, behaviour and lifestyle factors.

The HEALTHVENT project is developing health-based ventilation guidelines for non-industrial buildings in Europe (offices, homes, schools, nurseries, homes and day-care centres) to help avoiding investment and energy cost of ventilation systems driven by ventilation rates which are not supported by evidence of health, productivity and welfare benefits. The guidelines are planned to be available in 2013.

Another project, funded by the European Parliament, on "Schools Indoor Pollution and Health: Observatory Network in Europe (SINPHONIE)" is aiming to identify and analyse indoor air problems in schools and related health effects. The quality of ambient air, the impact of climate change and road traffic are also taken into account and assessed. The final report is planned to be published by the Commission in the first half 2013.

Summary

> Fine particles from outdoor origin are the largest single contributor to the overall burden of disease from indoor air pollutants in Europe.

Q5. What are the health effects of NO_2 in relation to the Air Quality Limit Values?

The current European Limit Values for NO_2 are equivalent to the WHO Air Quality Guidelines: 40 µg/m³ as an annual average, and 200 µg/m³ as a one hour maximum. These Limit Values are still regularly exceeded and questions have been raised about the health basis of these Guideline values. As part of the REVIHAAP project, WHO is currently addressing these questions in detail, and the answer provided is equivalent to the answer given by WHO, as there has been little recent EU funded research on health effects of NO_2 per se to discuss here. As shown earlier in this chapter, preliminary findings from the EU funded ESCAPE study have found NO_2 to be significantly associated with Natural Cause Mortality at levels well below the current annual Limit Value of 40 µg/m³. As shown in ESCAPE (Eeftens et al. 2012a; Cyrys et al. 2012), the spatial correlation between NO_2 and PM annual averages varies substantially across Europe and can be rather low in some areas. This confirms that NO_2 remains a useful indicator of air quality characteristics not fully captured by the PM-related markers.

In short term studies such as the EU funded APHEA 2 study, associations between NO_2 and mortality have been found to be independent of those with Black Smoke and PM10 (Samoli et al. 2006). There are only very few studies on long-term associations with NO_2 that have adjusted for particle mass. One example is a cohort study from the US which found NO_2 to be associated with mortality independent from PM10 (Hart et al. 2011) but the overall

evidence for a direct causal role of NO_2 in these associations is still limited. between NO_2 and mortality have been found to be independent of those with Black Smoke and PM10 (Samoli et al. 2006). There are only very few studies on long-term associations with NO_2 that have adjusted for particle mass. One example is a cohort study from the US which found NO_2 to be associated with mortality independent from PM10 (Hart et al. 2011) but the overall evidence for a direct causal role of NO_2 in these associations is still limited.

Chapter 2
Particulate Matter

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Introduction

The aerosol mass size distribution is characterized by two main modes: the accumulation (or fine) and the coarse modes (Figure 2.1). These modes have different chemical composition and sources. The fine mode originates from combustion-related sources and contains most of the particulate matter formed chemically in the atmosphere (secondary aerosols). The coarse mode contains mainly resuspended dust, sea-salt and other mechanically generated particles. With respect to the health effects of fine PM, these particles penetrate into our lungs more efficiently than the larger coarse particles that deposit mainly in the large conducting airways. Fine particle mass mainly comprises secondary inorganic species (ammonium, nitrate, and sulphate), elemental (or black) carbon and organic compounds, which can be classified into primary and secondary components. The contribution of the inorganic and organic fractions varies spatially and seasonally. Coarse particle mass is dominated by inorganic species, such as crustal minerals and sea salt. In addition to major components, fine and coarse PM contains also minor components that might be relevant in defining aerosol sources, and toxicological properties, such as black carbon transition metals, and aromatic or halogenated organic species.

Although many studies have addressed the question on effects of different PM sizes and composition over the last 15 years, it has not been possible to identify one or two chemical components that are clearly more harmful than others.

Epidemiological study tools have not so far been able to differentiate effectively the role of single PM components in health outcomes. However, health effects are clearly linked to PM mass concentrations, thus reducing PM concentration should provide an effective measure to reduce PM health effects.

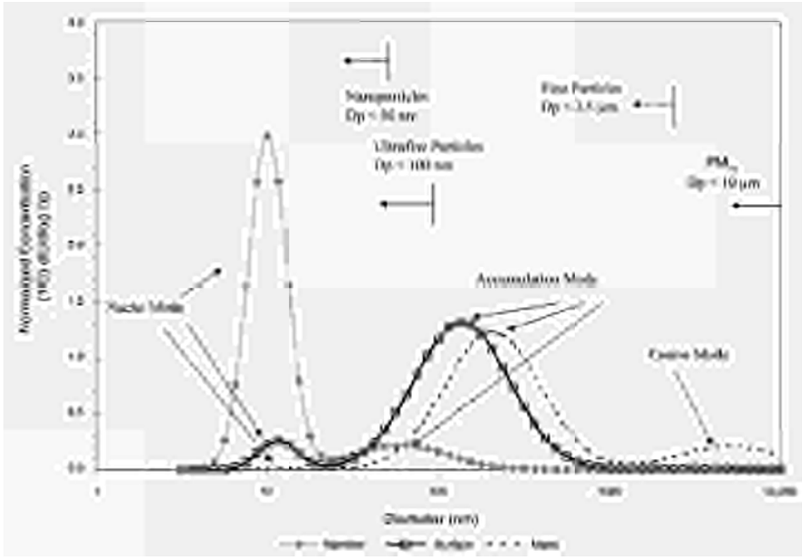

Figure 2.1 Normalized size distribution of particle number, surface, and mass for a typical roadside PM (HEI 2010).

Q1. PM size, what matters and which PM size fraction should be the focus of new regulatory effort?

Many studies link exposure to fine particles (PM2.5) to health effects (e.g. Pope and Dockery, 2006). The collective evidence from epidemiological, controlled human exposure and toxicological studies is sufficient to conclude that there is a causal relationship between short and long term exposures to PM2.5 and mortality and cardiovascular effects. The evidence for the health effects of coarse PM (PM>2.5) is more limited, but suggests relationships between short-term (but not long-term) exposure to coarse PM and cardiovascular effects, respiratory effects and mortality (e.g. Perez et al. 2008).

Scientific evidence indicates the need to regulate fine particles, and at the same time supports a link between coarse particles and health effects. Current standards focus on fine particles as PM2.5 and PM10 (coarse plus fine), thus the limit value that covers coarse particles is PM10, although this may not be the best way of controlling coarse particle sources. In the long term it may be necessary to regulate fine and coarse particles independently. The efficiency of measures aiming to abate specific emission sources would be evaluated more successfully using separate metrics for fine and coarse particles. In addition, fine and coarse particles are associated with different atmospheric residence time, and thus different travel distance, human exposure, (and climate effects). Thus investigations of the associations between PM and health effects would benefit from metrics which separate these particle fractions more clearly.

The existing knowledge on PM sources and chemical composition supports the use of the PM1 metric, instead of PM2.5, to separate the sources of fine and coarse PM. PM1 better represents the fine fraction than does PM2.5 as mass contributions of natural aerosol, such as mineral dust and marine aerosols, are reduced from PM2.5-10 to PM1 when compared to PM10 and PM2.5; in addition, chemical composition of the PM1-2.5 fraction is often similar to that of PM10 (Figure 2.2). Further evaluation of the PM1 metric seems useful. As also most of the mechanisms through which PM affects climate are related to submicron particles. At the moment there is limited epidemiological and toxicological evidence of PM1 health effects, to support such an approach.

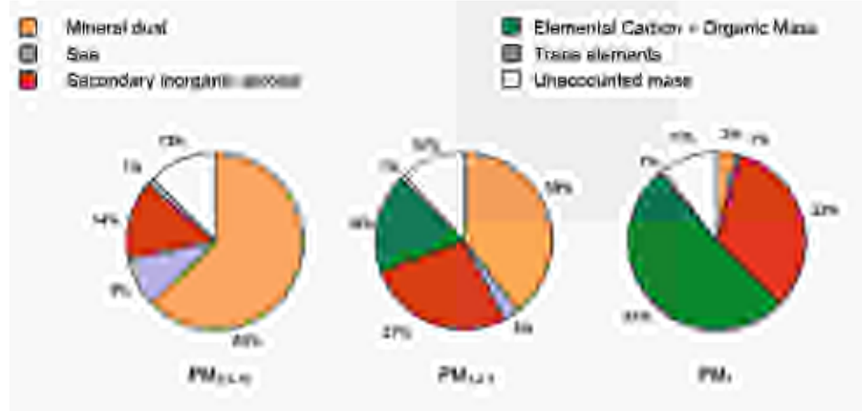

Figure 2.2 Mean annual contribution of major components to PM2.5-10, PM1-2.5, and PM1 in an urban background site in Barcelona. Adapted from Perez et al. (2010).

Summary

> Both fine and coarse particles are linked to health effects, although on different time scale exposure: fine particles effects are seen for both long and short term exposure, while coarse particles have been shown to have effects so far only from short term exposure.

> Regulation of PM2.5 and PM10 should be continued in the short term. In the long term quantitative knowledge of the link between coarse PM and health effect should be used to introduce independent regulation of fine and coarse PM.

> The metrics for fine and coarse particles, according to the present legislation, are PM2.5 and PM2.5-10, respectively. Potentially, PM1could be more effective in separating fine and coarse aerosols basedon chemical composition and sources although there will always be overlap between accumulation-mode and coarse-mode particles in the 1 – 3 µm size range. Furthermore, evidence of PM1 related health effects is limited.

Q2. Composition, do we know the link between aerosol composition and health effects?

Toxicological and epidemiological studies begin to reveal potential mechanisms responsible for PM health effects (WHO, 2007), such as

- obstruction of respiratory conducts and asthma;
- carcinogenic and carcinogenic response;
- oxidative stress and consequent tissue inflammation;
- translocation of nano-particles from the lungs to the circulatory system and from them to a number of organs.

The main chemical classes contributing to the health effects are expected to be soot (WHO, 2012), transition metals (Duvall et al. 2008) aromatic and halogenated compounds (Delfino et al. 2010), organic carbon (Hu et al. 2008) and primary biogenic aerosol particles - PBAPs (Polymenakou et al. 2008).

Soot is a component of fine PM, and is emitted by combustion engines (especially Diesel engines, with high EC/OC ratios), residential heating (including wood and coal burning, with lower EC/OC ratios), energy production through heavy oil and coal combustion, waste burning, and forest fires. Depending on the methodologies used for its determination, the light absorbing fraction of soot is quantified as BC (through optical measurements) or EC (through thermo-optical measurements). In particulate matter soot particles are associated to a large variety of organic species and metals co-emitted during combustion. Reviews of available epidemiological studies reveal a link between short-term variation of BC concentration and short term health effects, including all-cause and cardiovascular mortality and hospital admissions (WHO 2012). Cohort studies show a relation between BC exposure and long-term health effects, such as all-cause and cardiopulmonary mortality (Figure 2.3). The link between traffic-related health effects and BC concentration is more robust than with PM concentration, indicating that BC is a good indicator of traffic emission harmfulness. Nevertheless, review of toxicological studies shows that BC (or EC) might not be the major toxic component of PM, but may act as a carrier for more toxic substances emitted during combustion (WHO 2012). Instrumentation is now commercially available and affordable to implement BC (or EC) monitoring, but methodologies need to be standardized in order to facilitate result intercomparison. In addition to the considerable health benefit, soot measures would also help to evaluate the effectiveness of air quality plans implemented in zones with exceedance and high traffic-related emissions.

Laboratory experiments have shown that organic compounds in both gasoline fuel and diesel engine exhaust can induce oxidative response in cell or are associated with oxidative potential (Hu et al. 2008). In particular volatile organic species, likely secondary organic aerosol from condensation process, are characterized by the highest oxidative potential (Biswas et al. 2009).

A few epidemiological studies investigated the health effects of PM-bound transition metals. These studies observed a correlation between some transition metals (e.g. Ni, V, Cr, and As) and hospitalization for cardiovascular and respiratory disease, and cardiovascular mortality. Toxicological investigations link metals to increased particle oxidative potential and carcinogenic properties (Duvall et al. 2008). Finally, the health impact of metals was observed to depend on their solubility and bioavailability, which in turn is affected by particle age and processing (Harrison and Yin 2000).

Aromatic compounds, such as polycyclic aromatic hydrocarbons (PAHs), have been extensively investigated for their toxicity, mutagenic and carcinogenic effects. In addition, aromatic compounds can be converted into oxygenated species, like quinones, which induce oxidative stress response in cells.

PBAPs, such as viruses, bacteria, pollen, spores, fungi, plant and animal tissue debris, are associated with negative health impact, such as allergies and some diseases. PBAPs contribute to both fine and coarse PM and are present in rural and urban areas. Long distance transport of PBAPs, allergens, and pathogens has been proposed, for example, during Saharan dust outbreaks over Europe (Polymenakou et al. 2008). Combined exposure to pollen and air pollutants enhances allergic sensitisation, whereas it is still unclear whether this sensitisation arises from the impact of air pollution on the pollen on the exposed individuals.

Attention has been also dedicated to secondary inorganic aerosols (predominantly nitrates and sulphates), which make up a sizable fraction of PM2.5 in many areas (Rohr and Wyzga 2012 and reference therein). Epidemiological studies have frequently found these to be associated with adverse health effects. This is in contrast to findings from laboratory studies where exposure to higher than ambient concentrations of sulphates and nitrates generally fail to elicit much of a biological effect (Kelly and Fussel 2012). In ambient air, nitrates and sulphates possibly alter PM hygroscopicity, which may increase exposure to soluble transition metals and other toxic PM components.

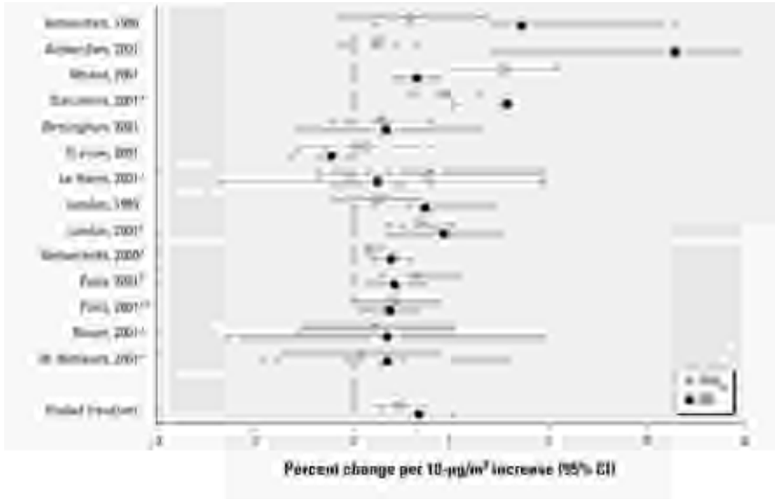

Figure 2.3 Single-city estimates of all-cause mortality for PM10 and black smoke BS (10 mg m^{-3} of BS correspond about to 1 mg m^{-3} of BC). Year indicates year of publication (Janssen et al. 2011 and reference therein)

Summary

> The air concentration of PM mass is the parameter that most closely links to health effects.

> There is suggestive evidence of a BC concentration link to short and long term health effects; although toxic mechanism has not been identified (toxic agent or carrier of other toxic substances), it would be useful to include it to the list of monitored/regulated pollutants. BC (or EC) might not be the major toxic component of PM, but may act as a carrier for more toxic substances emitted during combustion.

> A few classes of compounds in PM, representing a minor fraction of total mass, are known to induce relevant health effects: transition metals and aromatic organic compounds (PAHs). These compounds should be subject to ad-hoc regulation with time and space specific objectives.

> PBAPs are associated with negative health impact; although they have natural origin, their emission and transport can be affected, directly and indirectly, by anthropogenic activities.

> Recent studies show also the relevant health effects of primary and secondary organic aerosol, especially of the more volatile fraction.

> Secondary inorganic aerosol, which represents a large fraction of fine PM, has been linked to health effects by epidemiological studies, but not by laboratory studies.

Q3. What are the most important sources to control and in what order?

There is insufficient evidence to give firm recommendations on priorities for reduction of PM components so pressure should continue on all components of the PM mix, including both fine and coarse components. However there is emerging evidence, suggestive rather than definitive at this stage, that emissions from combustion sources, especially traffic, are potentially more potent than others, so that the regulatory process could consider giving priority to measures to reduce these emissions.

Traffic emissions contribute to local, urban and regional pollutant concentration, and traffic exposure is higher within distances up to 1500 m from highways and major traffic roads, where a large fraction of population resides. There is evidence that supports a causal relationship between exposure to traffic air pollution and exacerbation of asthma, and suggestive evidence of causal relationship with non-asthma respiratory symptoms, impaired lung function, total and cardiovascular mortality (HEI 2010). One of the surrogates used to evaluate traffic exposure is Black Carbon (BC) Effects may not be determined directly by BC toxicity, but toxicity of other species produced during combustion and associated with BC particles, such as organic molecules (e.g. PAHs and other aromatic compounds) and transition metals (WHO 2012).

Control precursors of secondary inorganic aerosol may be an effective measure to reduce total PM mass, although it may not be an optimal way of improving public health. Precursors of secondary inorganic aerosol are ammonia, nitrogen oxides, and sulphur oxide. For example, a 50% reduction in ammonia emissions is predicted to result in a 16% reduction of the PM2.5 levels in downtown London (Table 2.1). Ammonia is mainly produced by agricultural activities, including fertilizer use, waste management, and livestock, for which a wide range of effective control measures have been developed. In urban areas ammonia is emitted mostly by traffic and other fugitive sources such as city waste containers and sewage.

Reductions in nitrogen oxide emissions would also result in significant reductions in $PM_{2.5}$ levels; however they could be accompanied by increases in ozone levels in the same areas. For example, 50% reduction of nitrogen oxides would lead to 3 to 15% of PM2.5 reduction. Main sources of nitrogen oxides are energy production, industry, and traffic. Ambient sulphur dioxide and sulphate levels in PM clearly decreased in most of Europe during the last decades; a further reduction in sulphur oxide emissions will lead to a significant decrease of fine PM especially in the Eastern Mediterranean (Kulmala et al. 2011). In Athens, 50% reduction of sulphur oxides would reduce PM2.5 by 10%.

In addition to black carbon (or elemental carbon), organic carbon contributes substantially to fine PM mass, both in urban and rural locations. Primary organic carbon is mainly co-emitted with black carbon, thus BC control measures

would also lead to a reduction in OC. The emissions of fine primary organic carbon (OC) in Europe are dominated by traffic and residential combustion of wood and coal. In winter wood burning represents up to 20 to 30% of carbonaceous aerosol in urban areas where wood burning is common, and between 30 and 70% in some rural areas. Conversely, fossil fuel combustion is responsible for 10 to 40% and 40 to 50% of carbonaceous matter in rural and urban sites, respectively (e.g. Gelencser et al. 2007). The reduction of primary organic carbon emission sources would also result in the reduction of secondary organic aerosol precursors (volatile organic compounds VOCs and intermediate volatility organic compounds IVOCs), and thus would further contribute to PM abatement.

Resuspended road dust, tyre wear, brake and road pavement abrasion contribute to non-combustion traffic emissions, and are inorganic and organic compounds, including trace metals, that are also causing health effects. Non-combustion traffic emissions are not currently regulated although evidence suggesting a relationship with health outcome is increasing (see chapter 2).

Table 2.1 Simulated reduction in PM2.5 levels in major European cities for 50% emission reduction of different aerosol precursors (Kulmala et al. 2011)

	Reduction in PM2.5 (μgm^{-3}) for a Reduction of 50% Emissions of				
City	SO_2	NO_x	VOCs	POA	NH_3
London	1.1 (4.2 %)	2.9 (11 %)	0.8 (3.1 %)	0.7 (2.6 %)	4.0 (16 %)
Paris	1.3 (6.1 %)	1.4 (6.7 %)	0.2 (1 %)	1.9 (8.9 %)	0.8 (4.2 %)
Athens	1.0 (10 %)	0.4 (3.5 %)	0.1 (0.8 %)	0.2 (1.7 %)	0.6 (6.1 %)
Marseille	0.7 (3.7 %)	1.1 (6 %)	0.3 (1.3 %)	0.5 (2.5 %)	2.3 (12 %)
Ruhr	1.1 (10 %)	1.7 (15 %)	0.1 (1 %)	0.2 (1.8 %)	0.7 (6.4 %)

Summary

> Reduction of PM levels can be achieved by reduction of secondary inorganic aerosol through reduction of their precursors (ammonia, nitrogen oxides, and sulphur oxide), in particular, NH_3 emissions have been subject to very little control to date, including the Gothenburg revision. Technology solutions are now available to reduce NH_3 emissions, thus NH_3 emission control represent a valuable measure to control PM levels.

> Suggestive evidence, not definitive at this stage, points towards traffic emissions as potentially more potent than others for their health impact. An efficient surrogate of traffic-related emissions is Black Carbon (BC).

> Non-exhaust traffic emission is not regulated and is gaining special relevance, especially when exhaust emissions have been abated as in the last years. These emissions are a significant source of transition metals.

Q4. Hot spots or the background, which should be the priority for control of PM sources and precursors?

Definition of hot spot and background locations depends on the spatial scale; for example, megacities represent hotspots at regional level, while street canyons are local scale hotspots.
At the regional scale, the evidence indicates that air quality in hotspots and background are coupled. Air quality in urban areas can be strongly affected by regional transport of pollutants, as showed by long term studies of PM2.5 and PM10 trend over central and southern Europe (Figure 2.4). On the other hand, PM exceedance events are more frequently observed in urban areas located in high background regions.

A more general distinction is made differentiating among regional background, urban background, and urban hot spots. In Europe, urban background PM_{10} and $PM_{2.5}$ concentrations are about twice that of rural background, while the ratio between hotspots and urban background is about 1.5. Background areas should be the priority for controlling PM sources and precursors, since these areas are characterized by spatial and temporal representativeness high enough to evaluate the effect of implementation of air quality measures. Nevertheless, a significant fraction of the urban population lives in hot spot areas with higher exposures (HEI 2010). It follows that specific emission reductions are indicated for hot spots too, when differences with background concentrations are large, and when sensitive subgroups such as children and the elderly are exposed. Such measures include energy-efficiency modernization of old buildings (reduction of 40% of PM2.5 by 2030), use of district heating network (reduction of 60% of PM2.5 by 2030), and improvement of energy combustion plant efficiency of (up to 90% reduction of PM2.5 by 2030) (Kampffmeyer et al. 2011).

It is important to clearly define hotspot and background areas. In Europe the characteristics of an urban background may vary widely from the typical high density Mediterranean cities (were population lives, works and moves close to traffic) with the Nordic cities characterized by residential areas less exposed to traffic emissions (Viana et al. 2007).

Coming to the emission control, we should be aware that emission inventories are not homogenous on the temporal and spatial scale, and are not consistent across scales. This is not surprising, but it is now better quantified, (Butler

et al. 2008). For important health relevant pollutants like PM the discrepancy between down-scaled and bottom-up inventories for major cities can be up to a factor of 3-4. This has important consequences for: i) assessing the urban increment in pollution, ii) calculating related exposure, iii) providing policy advise on sources to be most prominently mitigated, and iv) predicting expected impacts of emission reductions. There is a need for further harmonization of emission inventories across Europe.

Finally, air pollution is a consequence of emissions sources not only at local, and regional level, but also at the global scale. In most cases controlling emissions at the local and regional scale is sufficient to reduce PM concentration, however, pollution is also transported on intercontinental scale, impacting regional air quality far from its emission sources (HTAP 2010). This makes it difficult for the countries to meet air quality targets, even after implementation of national emission reductions. This results in a need for a combination of local and regional emission control strategies together with international and regional cooperation.

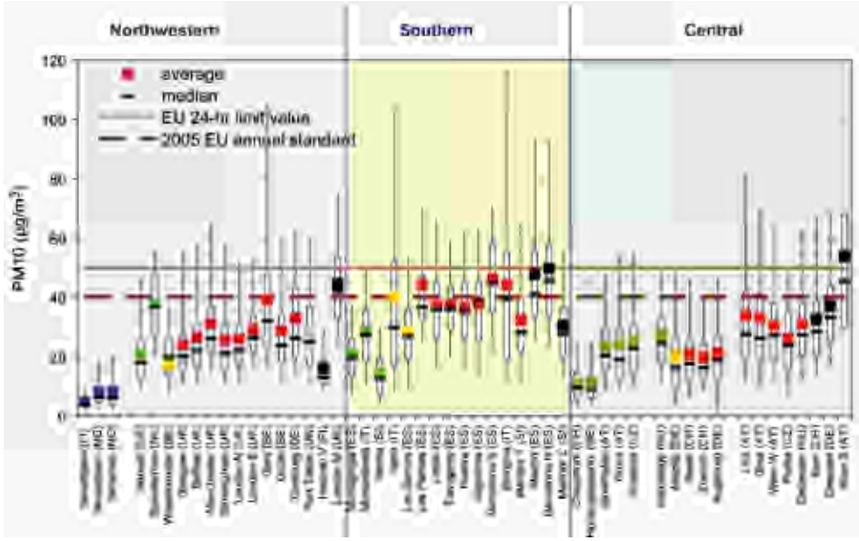

Figure 2.4 Annual average PM10 concentration in different geographical areas in Europe (Northwestern, Southern, Central) in remote (blue), rural (green), near city (yellow), urban background (red), and kerbside (black) sites (Putaud et al. 2010).

Summary.

> The greatest benefit of further control measures would be in the reduction of background PM concentrations: regional air masses affect urban areas where a large fraction of population is exposed, exceedance episodes are more frequent where high background PM levels are observed.

> Specific emission reductions (e.g increasing combustion plant efficiency, introducing district heating network, energy-efficiency modernization) are indicated for hot spots, especially when differences with regional background concentrations are large.

> Reduction of long-range transport of pollutants requires the control of emission sources at local, regional, and global scale. Regional and international cooperation is necessary to implement successful emission control strategies.

> Emission inventories are not homogenous on the temporal and spatial scale, and their consistency across scales needs to be improved.

Q5. Can the control of PM sources be regulated to maximise the benefits for health and climate effects?

PM pollutants, such as light absorbing species (BC) and light scattering species (sulphate and organic carbon OC) deteriorate air quality and contribute, with greenhouse gases, to climate change. Since the main sources of air pollutants are also sources of greenhouse gases, a synergic strategy should be adopted to mitigate climate change and improve air quality. Such a strategy should balance reduction of cooling and warming PM components, together with reduction of CO_2 emissions in order to meet climate targets.
A wide range of policy options for reducing emissions of air pollutants and greenhouse gases in urban areas has been investigated. The measure with the highest abatement potential for the whole European domain is the promotion of low emission vehicles. The corresponding abatement potentials are 6.8% for PM and 17.1% for CO_2 related to sectorial emissions (road transport) in 2030, and 0.7% for PM and 4.5% for CO_2 related to total emissions in 2030.

Concerning specific PM components, an important target is BC since it has both a clear warming climate effect and a negative impact on health. Recent studies demonstrate that for climate BC is a target as important as greenhouse gases, like CH_4 and CO_2. In addition, BC deposition on ice and snow in the Arctic is responsible for positive radiative forcing and Arctic atmospheric warming. BC emitted in and near the polar region has the greatest impact, but the Arctic

climate is also sensitive to extra-Arctic emissions (Hirdman et al, 2010).

A recent study initiated by United Nation Environment Programme (UNEP) and the World Meteorological Organization has investigated emission reduction measures for BC and ozone that can lead to an improvement in air quality while also benefiting the climate. A limited number of measures are necessary to reach co-benefits and they are based on control of black carbon emissions and co-emitted species, like methane (Shindell et al. 2012). The proposed solutions include technical measures, mostly end-of-pipe technology controls, for reducing emissions of incomplete combustion (diesel particle filters as part of a Euro VI package for road and off-road diesel vehicles) and also non-technical measures to eliminate the most polluting activities (elimination of high-emitting vehicles in road and off-road transport, including shipping; enforcement of an already existing ban of open field burning of agricultural waste).

Sulphate comprises a significant fraction of fine particle mass (10 to 20%) and is associated with negative radiative forcing (cooling). The implementation in Europe of SO_2 reduction measures during the last decades has led to a reduction in sulphate concentrations, and consequently to a warming effect. Such an effect depends on the reduction magnitude and the horizon over which the temperature change is assessed (Monks et al. 2009).

Chapter 3
Ozone

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Introduction

Tropospheric ozone is central to air quality as it is recognised to be a threat to human health (WHO 2003), have a deleterious impact on vegetation (Fowler et al. 2009) as well as being an important greenhouse gas (IPCC 2007) and also being key to the oxidative ability of the atmosphere (Monks et al. 2009).

Ozone remains a crucial air quality issue. There has been a substantial amount of EC funded work on ozone and its precursors in the urban environment in CityZEN and MEGAPOLI, and on the impact of transportation systems on air quality and on health (QUANTIFY, TRANSPHORM). The integrity of data and the ability to detect trends has been explored in the GeoMON project. International efforts, containing significant EU funded work, as part of UNECE HTAP (HTAP 2010) have assessed the state of the science with respect to the intercontinental transport of air pollutants in the Northern Hemisphere. Ongoing projects are focusing on the interaction between atmospheric chemistry and changing climate (PEGASOS, ECLAIRE).

The picture for ozone in Europe remains complex. The downward trend in ozone precursor concentrations has had differing effects on ozone in urban and rural regions. On the whole, the reductions in precursor emissions have not been reflected in decreased exposure to the detrimental effects of ozone.

There are a number of factors that could contribute to the lack of change in ozone.

a) *Background ozone*.
Observational evidence suggests that background ozone (see Box) concentrations over Europe are rising (Derwent et al. 2006; Jenkin 2008; Parrish et al. 2009; Simmonds et al. 2004; Wilson et al. 2012). The changing background contribution to European ozone levels could represent a substantial

future challenge to the attainment of ozone limit values (Derwent et al. 2010). With an increasing background contribution there is a requirement to control more ozone of anthropogenic European origin to achieve the limit (see Figure 3.1). Models have shown (Derwent et al. 2010; Szopa et al. 2006) that the benefit to European emission controls can be eroded by increasing background ozone.

What constitutes the tropospheric ozone background over Europe?

If all continental European man-made emissions of gases leading to the production of ozone (O_3) were switched off, there would still be ozone over Europe. That ozone is referred to as background ozone. It is derived from natural sources of ozone precursors in Europe, intrusions of ozone from the stratosphere, and from long-range inter-continental transport of ozone and its precursors formed from natural and anthropogenic sources in various regions of the world. European anthropogenic emissions generally lead to an increase of the ozone levels compared to the background levels. However, close to sources of nitrogen oxide (NO), ozone is reduced largely due to reaction of NO with O_3.

From (Hjorth and Raes, 2009)

Definitions

AOT40 - The seasonal accumulated exposure above 40 ppbV (80 µg/m^3) during daylight hours originally developed to assess this integral index. This is normally expressed as a cumulative exposure (ppbV h or ppm h).
SOMO35 - is the sum of the amounts by which maximum daily 8-hour concentrations of ozone (in µg/m^3) exceed 70 µg/m^3 on each day in a calendar year.

b) Interannual variability.
Ozone exhibits substantial interannual variability which may mask any change owing to emission reductions (Jonson et al. 2006; Wilson et al. 2012). Much of this variability is meteorological in nature (Andersson and Langner, 2007). Other sources of ozone variability include stratospheric-tropospheric exchange *e.g.* (Hess and Zbinden, 2011) and circulation pattern changes (Pausata et al. 2012). The inter-annual variability of O_3 is high (3.5-14 ppbV), making it difficult to detect trends (ca. 0.5 ppbV/yr) in urban O_3 from the relatively short records in urban and suburban monitoring networks. It is estimated that another 5 to 10 years of monitoring would significantly improve the signal (Colette et al. 2011; EEA 2009).

c) Long-range (hemispheric) transport of ozone and its precursors.
Substantial efforts have been undertaken as part of UNECE HTAP (HTAP

2010). Approximately 20 global models conducted idealised source-receptor experiments to quantify hemispheric transport of ozone and precursors . The key findings for ozone are reported in Fiore et al. (2009) and HTAP. The influence of 20% reductions of ozone precursor emissions (NO_x, CO, NMVOCs) from different world regions on monthly mean ozone, averaged across the EU region has been calculated. 20% reductions in North American, East Asian and South Asian emissions would reduce annual mean EU O_3 by about 0.4, 0.2 and 0.1 ppbV, respectively (HTAP 2010). The influence of LRT has a seasonality, with the largest impact in spring (March-April: ~0.8 ppbV) and minimum impact in late summer (July-September: ~0.45 ppbV) (HTAP 2010). These numbers may seem quite small, but to put them in context, the equivalent O_3 responses over the source regions are about 1.5-2 ppbV. Wild et al. (2012) found that the O_3 responses to emissions changes were linear within a certain range of perturbations; this indicates that the 20% responses can be safely scaled by actual changes in emissions from specific regions over short timescales. It is worth noting that the HTAP emission reductions (20%) have been exceeded in reality over the 1990-2009 period (EEA 2011). Wild et al (2012) also deconvolved the ozone trend over Europe from 1960-2000 into contributions from the EU, external sources (i.e. LRT) and the CH_4 trend. An increase of about 7 ppbV is calculated between 1960 and 1990, and a slight decrease from 1990-2000. Most of this trend came from non-European sources and CH_4, with a smaller contribution from changes in EU emissions.

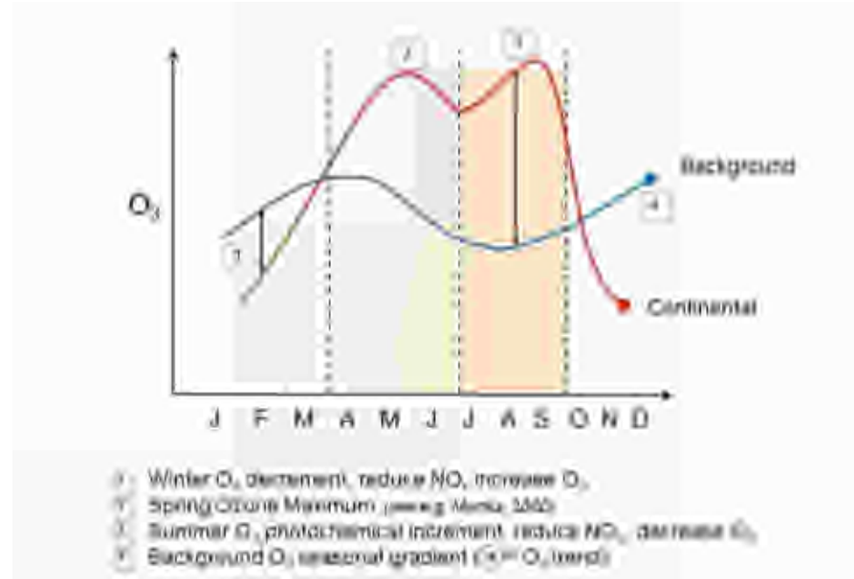

Figure 3.1 A schematic diagram of an idealized marine background and European continental ozone seasonal cycle (Wilson et al. 2012)

d) Shipping.
Nearly 70% of ship emissions occur within 400 km of coastlines, causing air quality problems through the formation of ground-level ozone, sulphur emissions and particulate matter in coastal areas and harbours with heavy traffic (Eyring et al. 2010). Offshore shipping sources of NO_x may become significant as land based-emissions decline (Dalsøren et al. 2010; Eyring et al. 2010; Jonson et al. 2009). The impact of ship emissions on tropospheric oxidants is mainly caused by the relatively large fraction of NO_x in ship exhaust. Dalsoren et al (Dalsøren et al. 2010) have shown that typical increases in yearly average surface ozone concentrations in the most impacted areas from shipping emissions are 0.5−2.5 ppbV. Shipping NO_2 emissions are visible from space and are related to economic cycles (de Ruyter de Wildt et al. 2012).

e) Stratospheric-tropospheric exchange (STE).
There is stratospheric contribution to background/trend/variability – e.g. Hess & Zbinden (Hess and Zbinden 2011). Ordonez et al (Ordonez et al. 2007) showed that the positive ozone trends and concentration anomalies in the lower free troposphere over Europe during the 1990s were probably due to enhanced stratospheric ozone contributions (dominated by changes in lower stratospheric ozone concentrations rather than by variations of cross-tropopause air mass transport), particularly in winter–spring. Similar findings have also been made by Tarasick et al (Tarasick et al. 2005) using the sondes network over Canada : "The long-term trends in average tropospheric ozone concentrations over Canada are similar to corresponding lower stratospheric trends, and tropospheric ozone levels show significant correlation with lower stratospheric ozone amounts."

f) Pyrogenic influence.
Wildfires/Biomass burning have the potential to be significant sources of ozone precursors (Jaffe and Wigder 2012). European fires in Portugal and Russia have been shown to contribute to air pollution (Konovalov et al. 2011; Martins et al. 2012) and the frequency is expected to increase with climate change (Carvalho et al. 2011). Further, long range transported biomass burning plumes can influence Europe (Cook et al. 2007; Real et al. 2007), though the impact is variable (Hudman et al. 2009).
g)Biogenic VOC emissions. There is much debate as to the influence at the European scale of biogenic VOCs in regional ozone production. Variation in natural plant emissions of ozone precursors could influence ozone concentrations. As stated in question one, model calculations have indicated that the biogenic isoprene emissions represent a major uncertainty in AOT40 with a factor of at least two (EEA, 2009). The impact of North American (NA) isoprene on European ozone has been assessed and it has been shown that future increases in NA isoprene emissions could offset decreases in EU surface O_3 resulting from controls on NA anthropogenic emissions (Fiore et al. 2011). Archibald et al have shown that there is a significant impact of mechanism uncertainties on the global impact of isoprene chemistry on ozone (Archibald et al. 2011). There is significant debate about the role of a wide range of biogenics in ozone chemistry under future climate e.g. (Andersson and Engardt 2010).

h) Emission inventories.
There are still very large uncertainties in anthropogenic emissions as represented in inventories. Though most global/regional/local inventories of anthropogenic emissions agree generally on the sign of the trends in the emissions over the past 2-3 decades, there are still very large inconsistencies between the levels of the emissions provided by these inventories (Colette et al. 2011; Granier et al. 2011).

Q1. How and can the background contribution to human health and environmental thresholds be quantified?

Much of the assessment of this question relies on the use of models. There are limited long-term observational datasets (e.g. Mace Head, Ireland) that can also be used directly for background ozone level assessment.

Model results have shown that changing ozone boundary concentrations on the western edge of Europe, have a relatively small influence on metrics such as AOT40 and SOMO35 in most parts of Europe (less than 10 %) (see Figure 3.2). In Scandinavia, the United Kingdom and western parts of the continent the influence could be larger up to 20% (EEA 2009). As noted previously, models have shown (Derwent et al. 2010; Szopa et al. 2006) that the benefit to European emission controls can be significantly counterbalanced by increasing background ozone.

Model calculations have indicated that biogenic isoprene emissions represent a major uncertainty in AOT40 with a factor of at least two, while less so for SOMO35 (EEA 2009).

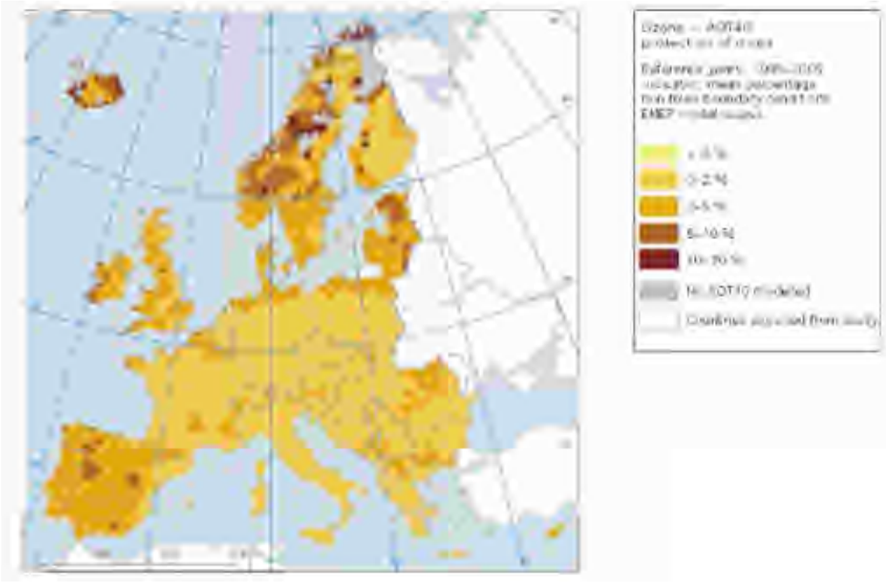

Figure 3.2 Absolute percentage differences in AOT40 simulated by the EMEP model when fixed boundary conditions are applied, rather than allowing them to vary inter-annually, for the period 1995-2006 (EEA 2009).

The AOT40, while still in use has been shown to lead to erroneous risk assessment and a flux based approach is preferred (Mills et al. 2011).

Ozone metrics are expressed in terms of the cumulative value of either stomatal fluxes or concentrations at the top of the canopy. Critical levels are preferably based on stomatal fluxes, as these are considered more biologically relevant since they take into account the modifying effect of climate, soil and plant factors on the uptake of ozone by vegetation. Annual mean O_3 reflects the impact of anthropogenic emission changes. However, it is not relevant for exposure assessment: annual mean O_3 is heavily influenced by wintertime and night-time low-ozone conditions, hence an upward trend does not necessarily reflect an importance for population exposure. For example, 90% of the SOMO35 value is produced during spring and summer (van Loon et al. 2007).

A recent WMO report (2011) recommended that for assessing possible shifts in the distribution of hourly average O_3 concentrations it is required to trend changes in the distribution of hourly average concentrations by month. Identifying statistically significant monthly changes in the mid- and low-level hourly average concentrations provide information for assessing changes in physical processes associated with global climate change, long-range transport, and the efficacy of models used for emission and risk reductions. In addition, for assessing how year-to-year variability influences trending patterns, it was

recommended characterizing moving 15-year trends for ozone. Owing to the differential changes that might occur over time to parts of the distribution, a careful selection of O_3 exposure metrics is recommended. For example, one exposure metric may focus on the higher concentrations in a distribution, while another metric might focus on the lower portion of the distribution; a different trending pattern might result between the two exposure metrics depending upon which part of the distribution changed over time (Lefohn et al. 2008, 2010).

Significant challenges remain in respect of modelling ozone at the regional scale across a range of process limits and metrics (van Loon et al. 2007). In general, threshold-based indicators are more robustly estimated by models when the threshold is low (Ellingsen et al. 2008)

Summary

> The contribution of background ozone to AOT40 or SOMO35 and the preferred flux based metrics for vegetation can be quantified but model uncertainty is large, partly because of uncertainties attributed to externalities such as precursor emission or long range transport in addition to the uncertainties of the photochemical models.

> Background ozone makes a relatively small contribution to the measures of peak exposure (AOT40 and SOMO35) but it dominates the mean ozone concentration and flux-based metrics. Background ozone contributes substantially to ground level ozone in the west of Europe and gradually becomes a smaller proportion of ozone towards the eastern countries.

> New proxies for damage from ozone exposure should be considered, as peak ozone may not be an effective measure of impacts on human health and on the environment, especially seasonally. In the case of ecosystem effects, the flux based methodology is a preferred metric and is measurable.

Q2. What is the contribution of background ozone to effects on human health and ecosystems?

In a health context, ozone is an aggressive and irritating air pollutant and the World Health Organisation (WHO 2007) characterise its human health impacts using the daily maximum 8-hour mean ozone concentration as a metric. Ozone levels in excess of this guideline are seen at almost every continental rural ozone monitoring station in almost every year (EEA 2011).

The adverse effects of peak ozone concentrations on the respiratory system have been well understood for several decades. More recently, short-term variations in ozone have also been shown to be associated with short-term changes in mortality. In comparison, much less is known about effects of long-term exposure to elevated background ozone concentrations on human health. Some studies have found effects on the development of asthma, and one major study has shown a relationship with respiratory but not cardiovascular mortality. No European studies have looked at long term effects of elevated background O_3 concentrations.

More details can be found in the chapter on air quality and health.

As part of HTAP (HTAP 2010), estimates were made of avoided premature mortalities from long-range transported ozone and its precursors. For each of the HTAP regions (North America (NA), Eastern Asia (EA), Southern Asia (SA), and Europe (EU)), HTAP found that the mortalities avoided that could be attributed to 20% emissions reductions in 'foreign' regions, as a percentage of those avoided by the same emissions reductions in all regions, were approximately 30%, 30%, 20%, and >50%, respectively (see Figure 3.3).

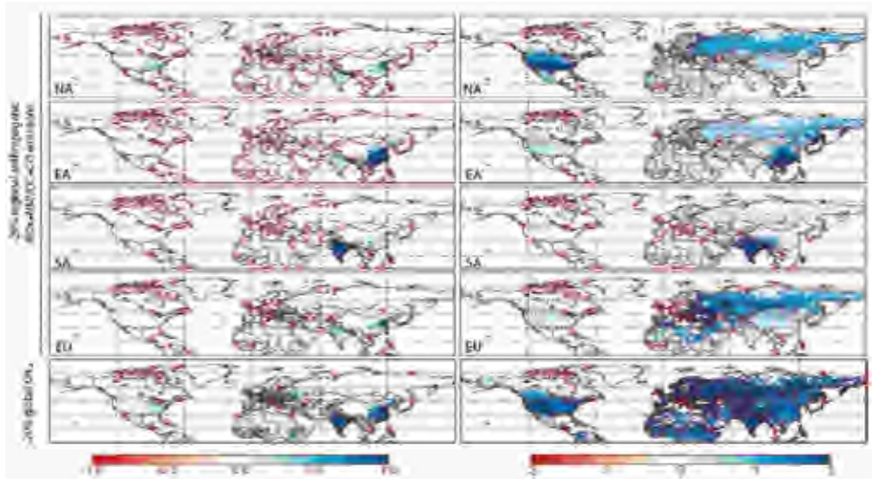

Figure 3.3 Annual avoided cardiopulmonary mortalities per 1000 km^2 (left) and per million people (right) from 20% NO_x, NMVOC, and CO reduction in the region shown and a 20% global methane mixing ratio reduction assuming no low-concentration threshold. (Anenberg et al. 2009)

That is, if emissions were reduced globally uniformly by 20%, over half of the mortalities avoided due to ozone in Europe could be attributed to non-European emissions. Reducing emissions in NA and EU avoids more mortalities outside the source region than within, owing in part to larger populations in external receptor regions. Lowering the global methane abundance by 20% reduces mortality in percentage terms most in SA, followed by EU, EA, and NA (Anenberg et al. 2009).

Recent work (Hollaway et al. 2012) has shown there is a significant impact of transboundary emission reductions on crop production e.g. a 100% reduction in N American anthropogenic NO_x emissions produces European crop production loss reductions of between 14.2% and 63.2 %. The threshold nature of the AOT40 ozone exposure metric results in strong dependence of non-local emissions impacts on the local ozone concentration distribution. Holloway et al (2012) state the their " … results demonstrate that local air quality and emission control strategies have the potential to partly alleviate ozone-induced crop yield loss in continents downstream, in addition to effectively mitigating local ozone-induced production losses".

Summary

> If ozone precursor emissions were reduced globally uniformly by 20%, over half of the mortalities avoided due to ozone in Europe could be attributed to non-European emissions

> Hemispheric emission reduction of ozone precursors can significantly reduce downwind crop production loss. For the case of Europe, reductions in anthropogenic NO_x emissions from N. America give the largest effect in crop production loss.

Q3. What is the relationship between control of NO_x and VOC sources in Europe and changes in ozone exposure and effects?

With respect to the ozone precursors, downward trends in VOC and CO concentrations have been observed in urban and rural areas (Derwent et al., 2003; vonSchneidemesser et al., 2010). For example, observed reductions in CO concentrations since 1999 (50 % at traffic stations, 35 % at urban stations and 25 % at rural stations) are in line with the reported 44 % reduction in total EU emissions of CO (EEA 2011). Work in CityZen showed that in London and Paris the biogenic VOC isoprene's importance for O_3 formation has not increased despite hydrocarbon concentration reductions (vonSchneidemesser et al. 2011). Monitoring isoprene only as a proxy for biogenic influence may not be sufficient, as the group of biogenic VOCs that are ozone precursors are much wider. The situation with respect to NO_x and in particular NO_2 emissions is less clear. From 1990 to 2008, the European Environment Agency reported that NO_x emissions from road transport in the EU-27 have decreased by approximately 40% with a virtually constant gradient to 2008. However, it is now clear that ambient concentrations of NO_x (NO_x=NO+NO_2) and NO_2 have not declined as fast they would do in response to reported emission reduction and some

locations show little or no downward trend. This has led to a large number of EU Member States failing to meet the annual average limit value for NO_2 of 40 µg m^{-3} in Directive 2008/50/EC. The principal problem areas are busy roads, so the levelling off of road transport NO_x is a matter of some concern (Carslaw et al. 2011). There remains policy challenges for NO_x in trade-offs between the certainty in terms of health impacts of limit values and the cost of mitigation. (Williams and Carslaw 2011). Figure 3.4 shows the impact of transport emissions on ozone in Paris.

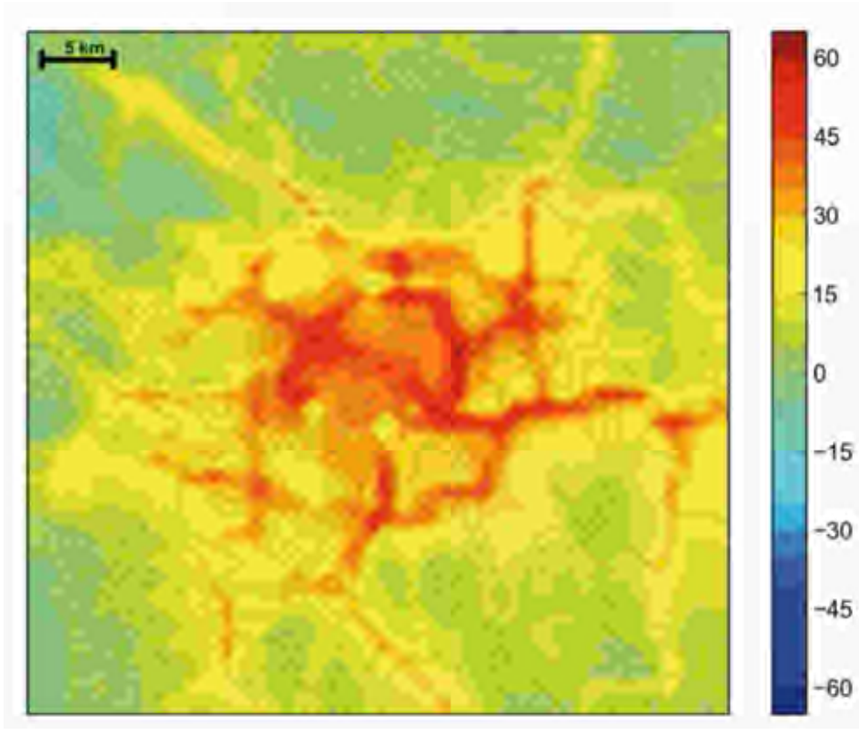

Figure 3.4 Difference map for O_3 concentrations (units are mg m^{-3}) in the Paris metropolitan area, calculated with baseline and the zeroed out traffic emissions (from MEGAPOLI Pandis et al. 2010).

A model-based comparison of simulations where anthropogenic emissions are kept constant has been undertaken as part of CityZEN. It was found that the magnitude of the emission-driven trend exceeds the natural variability for the primary precursors and therefore concluded that emission management strategies have had a significant impact over the past 10 years, hence supporting further emission reductions (Colette et al. 2012a) EMEP4UK (and other models) show as NO_x emissions are reduced, ozone generally increases, at least within the NO_x-saturated parts of Northern Europe (see Figure 3.5) and the widespread decrease over NO_x limited areas in Southern Europe. It may be more relevant to discuss total oxidant in this context.

Keuken et al (2009) have argued that owing to the importance of the ozone/NO_x equilibrium local NO_x emissions would need substantial reduction to achieve lower NO_2 urban background levels. MEGAPOLI work has shown that mega-city city-based abatement strategies will not substantially impact regional ozone.

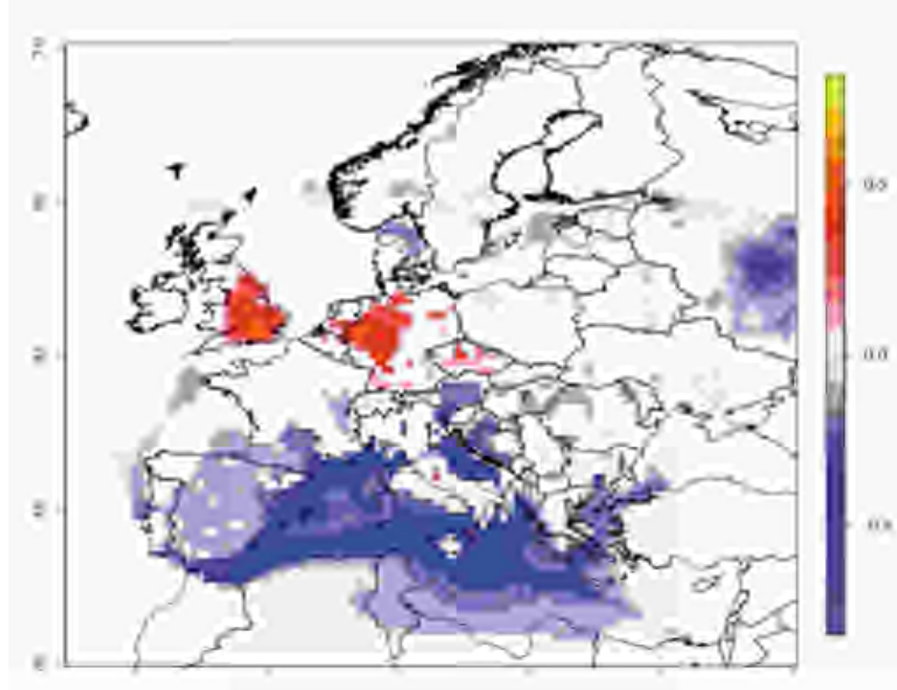

Figure 3.5 Ensemble model mean trend of ozone owing to the change in anthropogenic emission evolution alone (Colette et al. 2011).

Summary

> Within the NO_x-saturated parts of Northern and Western Europe and within urban areas, ozone generally increases when NO_x emissions are reduced. It may be more relevant to discuss total oxidant in this context.

> Abatement strategies focusing on megacities will not substantially impact regional ozone; there is a need for more than a city-based ozone precursor reduction policy.

Q4. How much benefit would control of methane deliver in mitigating ozone effects in Europe?

Methane is a relatively long-lived ozone precursor gas and has an important influence on *background* ozone levels. The globally averaged abundance of methane has increased by a factor of 2.5 from its pre-industrial value of about 715 ppbV (Etheridge et al. 1998) to a value of 1794 ppbV in 2009 (Dlugokencky et al. 2011). This increase, proportionately far greater than the parallel increase in CO_2, is driven mostly by increased anthropogenic emissions from fossil fuels, industry, agriculture (including biomass burning) and waste, but changes in CH_4 lifetime and meteorological feedbacks may also have played a role. With an atmospheric lifetime of about 10 years, methane is distributed relatively uniformly through the troposphere and thus acts as a source of ozone on a global scale, with little dependence on the location of its terrestrial source (Fiore et al., 2008). Model attribution studies suggest that increases in methane since 1960 are responsible for about one third of the increase in global mean surface ozone (1.4 ppbV out of 4.0 ppbV), and for a similar proportion over Europe (1.8 ppbV out of 5.6 ppbV, Wild et al.2012) (see Figure 3.6). As these are changes in background ozone, increases in methane will affect rural ozone across Europe.

There are shortcomings in the methane emission inventories for Europe (Bergamaschi et al. 2010) but it is clear that Europe makes a relatively small contribution to global methane budgets (Bergamaschi et al. 2009). Estimates of all anthropogenic sources over Europe from the EDGAR v4.2 emissions database are about 30 Tg/yr. Based on the ozone responses derived by Fiore et al., 2008, eliminating these emissions entirely would produce an average surface ozone reduction of only about 0.4 ppbV, with this impact distributed globally. The benefits of regional emission reductions for regional ozone control are thus relatively small. Given the nature of methane emissions it is likely that global policies are required to reduce methane (Shindell et al. 2012), but the benefits of these will be reductions in surface ozone that will be felt globally.

Methane reduction has potential win-win benefits for air quality and climate (Shindell et al. 2012). Studies have revealed the increasing importance of limiting atmospheric methane growth as emissions of other precursors are controlled, but highlighted differences in modelled ozone responses to methane changes of as much as a factor of two, and this uncertainty indicates weaknesses in our understanding and representation of the controlling chemical and dynamical processes in current models (Wild et al. 2012). Work by Fry et al (Fry et al. 2012) has shown that CH_4, CO, and NMVOC reductions reduce climate forcing better than NO_x reductions.

Summary

> Methane reduction has potential win-win benefits for air quality and climate

> Substantial benefits for ecosystem and human health result from large reductions in global methane, but the effect of controls in Europe alone would be relatively small.

> Ensemble modelling shows that trends of annual mean ozone attributed to the mitigation of anthropogenic emissions is masked by natural variability over most of Europe, except over the greater UK-Benelux-Germany region as well as Moscow and the Mediterranean.

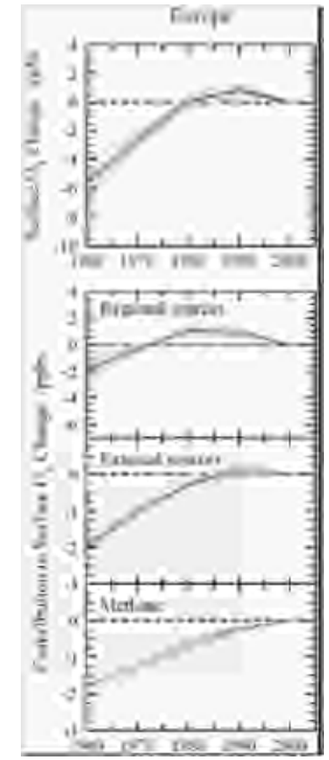

Figure 3.6 Modelled changes in annual mean surface O_3 over Europe from 1960 (in ppbv referenced to 2000 levels) due to anthropogenic precursor emission changes and changes in atmospheric CH_4. Coloured line shows the ensemble mean response from 14 models contributing to the HTAP model intercomparison, grey lines show individual models. The lower panels show attribution to European emissions, anthropogenic emissions elsewhere in the world, and changes in atmospheric CH_4 abundance (Wild et al. 2012).

Q5. Can the control of ozone precursors in Europe be regulated to maximise the benefits for health and climate effects?

Addressing air pollution and climate change together provides a unique opportunity to simultaneously achieve both air quality and climate policy goals in the mid-term (<50 years). Air pollution regulations are an important component of strategies to protect human health and ecosystems, including crop production. However, some air pollution emission controls, such as sulphur reductions, have adverse effects on climate change mitigation efforts. Equally, measures to mitigate climate change can have adverse effects on air quality.

Therefore, simultaneous solutions that offer net benefits for both air quality and climate taking into account the possible trade-offs between human health, food and water security and ecosystems would be advantageous.

An assessment of existing emission projections of greenhouse gases and air pollutant (IPCC RCP, Global Energy Assessment, and European Commission Roadmap for moving to a low carbon economy by 2050) shows that the magnitude of the co-benefits of climate policies for air pollution varies depending on the projections. But none of the scenarios investigated yields a negative trade-off i.e. increased air pollutant emission with GHG mitigation scenarios (Colette et al. 2012). In terms of impacts, average annual O_3 increases in some NO_x-saturated urban centres in the climate projections, but again the comparison of the reference and mitigation pathways shows that O_3 air pollution decreases substantially when the climate policy is enforced. It is worth noting that in general, changes in surface O_3 due to climate change are much smaller than those from anthropogenic emission reductions over the same time period (Langner et al. 2012).

Summary

> Integrated assessment of air pollution and climate policy synergies are a vital part of the process.

> Ozone air pollution decreases substantially when a climate policy designed to limit global warming to 2°C by the end of the century is enforced.crop yield loss in continents downstream, in addition to effectively mitigating local ozone-induced production losses".(from MEGAPOLI Pandis et al. 2010).

Chapter 4
Nitrogen

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Introduction

The European nitrogen cycle and the effects of reactive nitrogen on climate and the environment have been examined in detail over the last five years under the lead of the NitroEurope Integrated Project, funded under the 6th Framework Programme (www.nitroeurope.eu; Sutton and Reis 2011). NitroEurope provided the central foundation to conduct the work of the European Nitrogen Assessment (Sutton et al. 2011b). Together with input from the Nitrogen in Europe (NinE) and COST 729 programmes of the European Science Foundation, the findings have provided input to support the UNECE Convention on Long-range Transboundary Air Pollution (CLRTAP), especially through the coordinating activities of its Task Force on Reactive Nitrogen (Sutton et al. 2011a,c; Reis et al. 2012; TFRN 2012). These issues are now being extended in the ÉCLAIRE, examining how climate change may alter the air pollution threat to ecosystems (ECLAIRE 2013), thereby making a contribution to addressing the emerging challenges related to nitrogen (Sutton et al. 2009c).

Many of the major outcomes, as addressed in the following sections, have been discussed in detail in the ENA, with the key aspects collated from the ENA Summary for Policy Makers. Individual chapters of the ENA, which give a more comprehensive picture, have been referred to and are accessible online.

Q1. How do nitrogen emissions affect air quality?

Nitrogen is an extremely challenging substance for air quality: it is present in the air in many different chemical forms and it has proved hard to achieve substantial reductions in its emissions. The first point to recognize is that 78% of the world's atmosphere is nitrogen, present in the form of di-nitrogen gas (N_2). This is the stable state of nitrogen, which because of its unreactive nature provides a necessary stability to our atmosphere that moderates other changes. However, atmospheric N_2 can be converted into 'reactive nitrogen' compounds (collectively called N_r), forming both oxidized nitrogen compounds (NO_y) and reduced nitrogen compounds (NH_x). It is the emission of these reactive nitrogen compounds that generates the problems for European air quality.

There are many different forms of reactive nitrogen present in the environment, and it is important to see the atmospheric component as part of the wider nitrogen cycle. In particular, as Figure 4.1 illustrates, NO_y emissions are dominated by high temperature combustion processes, such as in large combustion plants and vehicle engines, while NH_x emissions are dominated by agricultural activities, especially those including livestock. The NO_y and NH_x react with each other and other atmospheric constituents to form nitrogen containing aerosol. These aerosol undergo long-range transport in the atmosphere, with the different N_r forms produced eventually being removed by a combination of direct uptake by the ground ('dry deposition') and scavenging by clouds and precipitation (leading to 'wet deposition'). These N_r compounds contribute to high nitrogen dioxide (NO_2) concentrations in urban areas, fine particulate matter (PM2.5 and PM10) and tropospheric ozone (O_3) concentrations, each of which pose a threat to human health (Hertel et al. 2011; Moldanova et al. 2011). In addition, with parallel emissions of nitrous oxide (N_2O) from soils and industry, N_r poses a risk for climate change and depletion of the stratospheric ozone layer (Butterbach Bahl et al. 2011; Ravishankara et al. 2009). Exposure of ecosystems to high N_r concentrations and the reaction products, such as tropospheric O_3, as well as their deposition leads to both phytotoxic effects and long term changes in natural and semi-natural ecosystems. These include crop losses due to enhanced O_3 concentrations, increases in forest productivity due to N_r deposition (which may partly offset the warming effects of N_2O by removing more CO_2 from the atmosphere), and loss of terrestrial biodiversity, as plants characteristic of natural ecosystems are out-competed by aggressive competitor species (Dise et al. 2011).

Although this represents only a short-list of the effects of N_r in the atmosphere, the complexity of its many effects should already be becoming clear. Figure 1 therefore provides a helpful summary of the main N_r forms and the associated environmental effects. In addition to effects on air quality and climate, it should also be noted that much N_r is lost directly to surface and ground waters,

creating problems of eutrophication, especially in coastal zones (Grizzetti et al. 2011). A key message is therefore that better overall management of the nitrogen cycle has the potential to improve nitrogen use efficiency (NUE) of the full system (from N_r formation to ultimate products), while simultaneously reducing losses and pollution of the environment (Sutton et al. 2011c).

In the following sections, effects are considered that address both the NO_y and NH_x components in more detail. The atmospheric cycle of NO_y is highly complex, but has received the most attention from both scientists and policy makers. Under very high temperatures in the presence of oxygen, N_2 is converted to nitric oxide (NO), which rapidly reacts with other atmospheric oxidants such as ozone to form nitrogen dioxide (NO_2), the combination being terms nitrogen oxides (NO_x). Subsequent transformations form both nitric and nitrous acids (HNO_3, HNO_2), particles (such as ammonium nitrates and ammonium sulphates) and a complex suite involving literally hundreds of different oxidized organic nitrogen compounds. Many technologies have been developed and increasingly adopted to reduce NO_x emission, mainly focusing on catalytic and non-catalytic reduction of NO_x back to N_2, as used in large-combustion plants and in catalytic converters on vehicles. Offset against these achievements has been a steady increase in vehicle use, so that net NO_x emissions have not decreased as much as had been hoped over the last 20 years.

If most attention has been given to NO_x emissions, by far the largest driving change to the European nitrogen cycle is the industrial reduction of atmospheric N_2 to form ammonia gas (NH_3), which is used to make fertilizers and many industrial products. This process fixes around three times as much atmospheric N_2 into N_r in Europe than is achieved by oxidation to NO_y compounds, accounting for 11.2 Tg N_r formation per year. The fertilizer N_r entering Europe's agricultural systems, together with a further 1.3 Tg from biological nitrogen fixation is cycled through plant, animal and organic matter, with a substantial fraction being lost to the environment. For air quality, the key threat is atmospheric NH_3 emissions from livestock farming systems and from mineral fertilizers such as urea, which amount to around 3.2 Tg N_r per year across the European Union. These ammonia emissions constitute a threat to European biodiversity, dispersing fertilizing N_r onto natural and semi-natural ecosystems, form particulate matter, constituting a threat to human health, and provide a vector for subsequent increased emissions of N_2O. In addition, although smaller, at 0.15 Tg N_r per year, European agricultural and others soils are a significant source of NO_x emissions (Leip et al. 2011).

For ammonia (NH_3) the first policy challenge has simply been to recognize the problem as an international concern. This recognition was effectively achieved in 1999 with the inclusion of NH_3 as a regulated pollutant under the UNECE Gothenburg Protocol, with emissions ceilings for NH_3 subsequently being adopted under the National Emissions Ceilings Directive. However, few

reductions in NH_3 emissions have been achieved, with very limited further commitments under the recently revised Gothenburg Protocol (Reis et al. 2012). Although a few countries like the Netherlands and Denmark have shown the it is possible to reduce NH_3 emissions through active mitigation policies, while retaining profitable agriculture, most other countries in the EU have so far not followed with similar policies (Bleeker et al. 2009; Oenema et al. 2011). **As the ENA and TFRN have shown, and is discussed further below, the economic costs of damage due to NH_3 greatly outweigh the modest cost of abatement measures, and significant emission NH_3 reduction could be achieved from a technical perspective.** The barriers to change appear to include both social and political dynamics and new approaches are therefore needed that recognize and address these constraints.

One of the key constraints for better understanding of the interactions between oxidized and reduced N_r forms for air quality is the linearity of relationships to particulate matter formation. The message of current studies is that to achieve a substantial reduction in PM2.5 concentrations across Europe will require significant reductions in both NO_x and NH_3 emissions.

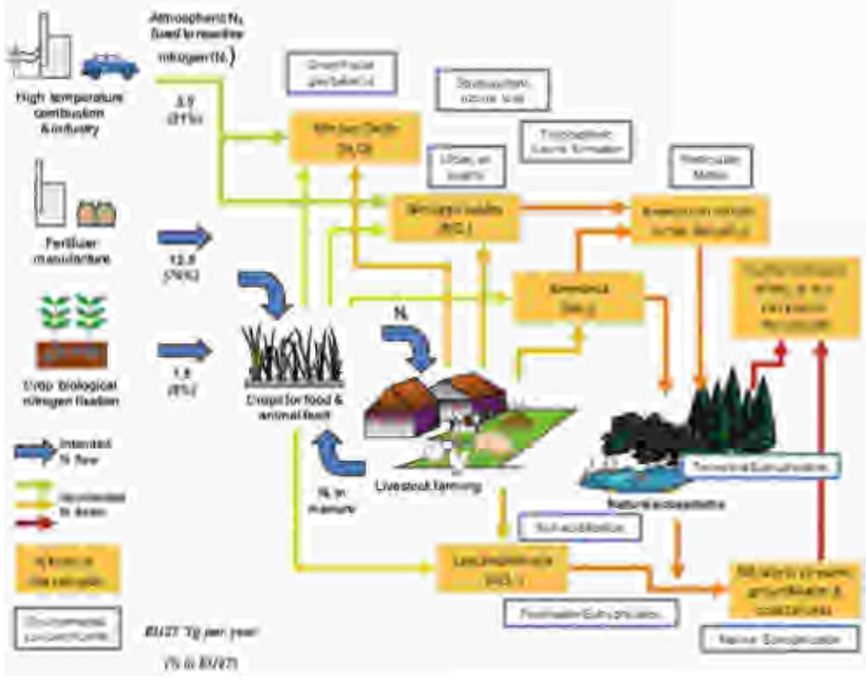

Figure 4.1 Summary of the main intended and unintended nitrogen flows, forms and their consequences for air quality and other environmental threats. Atmospheric di-nitrogen (N_2) is fixed to form a suite of interacting reactive nitrogen (N_r) compounds, associated with a wide range of environmental concerns (based on Sutton et al. 2011b).

Q2. How do the effects of nitrogen emissions on climate change interact with air quality?

Emissions of N_r have both warming and cooling effects on the global climate. The main warming components are increasing concentrations of nitrous oxide (N_2O) and tropospheric ozone (O_3), which are both greenhouse gases. The main cooling effects are atmospheric N_r, which is deposition presently increasing CO_2 removal from the atmosphere by forests, and the formation of N_r containing aerosol, which scatter light and encourage cloud formation see Figure 5.2 (Sutton et al. 2009b; Butterbach-Bahl et al. 2011). In addition, Ravishankara et al. (2009) have identified N_2O as a key contributor to stratospheric ozone depletion, reinforcing the relevance of tackling N_2O not only for its contribution to radiative forcing.

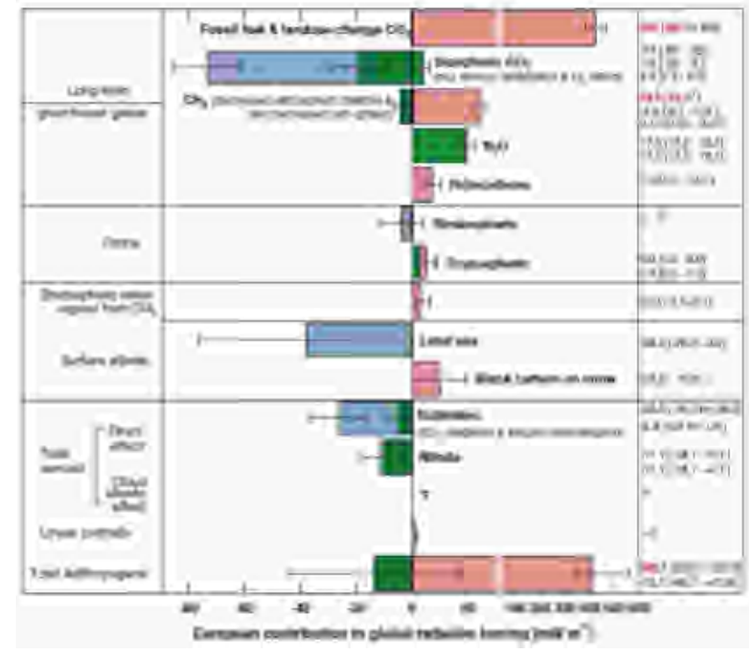

Figure 4.2 Estimate of the change in global radiative forcing (RF) due to European anthropogenic reactive nitrogen (N_r) emissions to the atmosphere.
Red bars: positive radiative forcing (warming effects); **light green bars**: positive radiative forcing due to direct/ indirect effects of N_r; **blue bars**: negative radiative forcing (cooling effects); **dark green bars**: negative radiative forcing due to direct/indirect effects of N_r. For biospheric CO_2, the dark green bar represents the additional CO_2 sequestered by forests and grasslands due to N_r deposition, while the light green bar represents the decrease in productivity due to effects of enhanced O_3 caused by NO_x emissions. For CH_4 the positive (not visible) and negative contributions represent the effects of N_r in reducing CH_4 uptakes by soil and the decreased atmospheric lifetime, respectively. Other contributions include the positive effect of tropospheric ozone from NO_x and the direct and indirect cooling effects of ammonium nitrate and sulphate containing aerosol (Butterbach-Bahl et al. 2011).

European N_r emissions are estimated to have a present net cooling effect on climate of −16 mW per m^2, with the uncertainty bounds ranging from substantial cooling to a small net warming (−47 to +15 mW per m^2). The largest uncertainties concern the aerosol and N_r fertilization effects, and the estimation of the European contributions within the global context. It should be noted however, that the N_r effect on biospheric CO_2 update may tend to saturate over future decades, while the longest term effect is from N_2O, due to its long residence time in the atmospehere. Therefore, while present N_r pollution may have net cooling effects, there is an expectation of long-term commitment to net warming effects.

There are many opportunities for 'smart management', increasing the net cooling effect of N_r by reducing warming effects at the same time as other threats, e.g., by linking N and C cycles to mitigate greenhouse gas emissions through improved nitrogen use efficiency. Measures that reduce emissions of NH_3, NO_x, N_2O and N_2 and reduction of N_r leaching all promote an increase in nitrogen use efficiency, such as in food and for bioenergy production. This means that less new N_r inputs (from fertilizers and biological nitrogen fixation) are needed to produce the same amount of food, thereby providing potential to reduce all forms of N_r pollution.

Summary

> Nitrogen pollution effects on climate, include warming from N_2O and the N contribution to tropospheric ozone, and cooling from the N effect on biosphere CO_2 exchange and from N_r containing aerosol

> Overall, a net cooling effect is estimated for present emissions, though the warming effect of N_2O is the longest term consequence, while the cooling effect of CO_2 uptake may tend to saturate over future decades.

> Strategies to reduce the warming effects of N_r need to reduce both N_2O and NO_x emissions, as well as NH_3 emissions, which contribute to indirect N_2O sources.

> Efforts to improve nitrogen use efficiency (NUE), including improved techniques in agriculture and reduction of all forms of N_r loss, allow more food to be produced with less new N_r inputs.

> Improving NUE must be a central element in strategies to reduce emissions of N_2O, NH_3 and NO_x simultaneously, demonstrating a clear synergy between air quality and climate policies. Better N management for air quality will therefore simultaneously deliver reductions in N_2O emissions.

Q3. Could ecosystem effects of NH_3 and the potential of a new AQ limit value be used in delivering Habitats Directive commitments?

In the frame of the COST Action 729 on *Assessing and Managing nitrogen fluxes in the atmosphere-biosphere system in Europe* a workshop brought together scientists, environmental managers and policy makers to address the threat of nitrogen deposition to the Natura 2000 network (Hicks et al. 2011). It was concluded that existing legislation controlling N_r emissions to air does not adequately address the impacts of nitrogen on the Natura 2000 network, and that there was need for much closer linkage between future air quality policies and the Habitats Directive.

In particular, the workshop recognized that the Habitats Directive adopts a precautionary approach for the proection of Special Areas of Conservation (SACs), which represent the flagship network of protected sites for the conservation of European natural habitats. This is illustrated by Article 6(3), of the Habitats Directive, which states that:

"Any plan or project … shall be subject to appropriate assessment of its implications for the site in view of the site's conservation objectives." … " the competent national authorities shall agree to the plan or project only after having ascertained that it will not adversely affect the integrity of the site concerned…."

There many plans and projects that may adversely affect Natura 2000 sites through by increasing atmospheric nitrogen pollution levels and deposition. In the case of large combustion plants and other point sources of NO_x, it was noted that a strong regulatory framework is already in place, as is the case for activities regulated under the Industrial Emissions Directive (IED). The IED includes large pig and poultry farms above certain thresholds, so to this extent a regulatory framework to protect SACs is also in place for agricultural NH_3 sources. However, only around 20% of European NH_3 emission arises from IED regulated sources. For the 80% of European NH_3 emissions, there is typically insufficient regulatory procedure in place, with the result that relevant plans and projects are typically not assessed in regard to their possible impact on the Natura 2000 network. This includes the development of many cattle and pig farming activities.

The workshop also noted that there was a need to develop a common approach to assessing N_r deposition impacts on individual Natura 2000 sites, and on the conservation status of habitats and species has been proposed, including reliable information on stock at risk, evidence of recovery, and the restoration potential. Future approaches could build on established methods such as the critical loads assessment for the 2013 reporting round under Article 17 of the Habitats Directive and for assessments under Article 6.

The workshop evaluated possible policy options that could provide tools to meet the existing commitment to protect the Natura 2000 network more effectively

(Sutton et al. 2011d). One of the options in particular, was to consider the potential of establishing an air quality limit value for NH_3 over the domain of Natura 2000 sites. In this regard, it is noted that while there are AQ limit values for many pollutants (including NO_2, SO_2, O_3, PM , etc.), the question of setting a limit value for NH_3 has, until now, been neglected.

'Critical levels' for NH_3 have recently been revised as part of the NitroEurope activity (Cape et al. 2009; Sutton et al. 2009a) and these provide a scientific basis describing the NH_3 air concentration above which adverse effects on species and habitats are expected to occur according to current knowledge. In the case of higher plants the critical level was set at an annual mean of 3 (2-4) mg m^{-3}, while for lichens, bryophytes and habitats where these are important to ecosystem integrity, the critical level was set at an annual mean of 1 mg m^{-3}. The workshop concluded that these critical levels could provide the starting point for discussion on setting an air quality limit value. The values themselves have now already been adopted by the UNECE Convention on Long-Range Transboundary Air Pollution (UNECE 2007).

By setting the critical levels as annual means, the UNECE also allowed that assessment avoid the need for expensive continuous monitoring of NH_3 concentrations. Rather, time-integrated monitoring (e.g. monthly values), using either active or passive methods provides a low-cost approach to assessing whether the annual NH_3 critical level is exceeded.

It is well established that NH_3 concentrations are highly spatially variable, decreasing rapidly in the first 1-2 km from pollution sources before being dispersed to the wider atmosphere. This means that there is substantial potential for local measures (such as buffer zones avoiding emissions, use of woodland belts, or requirement for local mitigation techniques) to reduce NH_3 concentrations in the vicinity of Natura 2000 sites. Further analysis is required, but initial analyses indicate that such spatial approaches could provide a highly cost effective local complement to the National Emission Ceilings that helps Europe meet its existing commitment under the Habitats Directive to protect that Natura 2000 network.

Summary

> Ammonia (NH_3) represents a priority pollutant for future European mitigation strategies, given its substantial contribution to the exceedance of critical loads and critical levels, and the limited extent of policies so far implemented.

> Further reductions in emissions of N_r emissions and NH_3 in particular are required to reduce effects of N_r on the Nature 2000 network, which is currently estimated to be under significant risk of nitrogen deposition, leading to changes in species composition with loss of key elements of biodiversity.

> The critical levels for NH_3 have been revised with the support of FP6 research and are now adopted by the UNECE CLRTAP. These values are set based on annual means, and can therefore be compared with measurements using current low-cost monitoring techniques. The critical level could provide the starting point for considerations to set an AQ limit value for NH_3.

> The establishment of an AQ value for NH3 could provide substantial benefits for meeting existing commitments to protect the Natura 2000 network of Special Areas of Conservation, designated under the Habitats Directive. This would foster the adoption of local planning and mitigation measures related to NH3, which would provide a cost-effective complement to the emissions reduction expected from revising the National Emissions Ceilings.

> Given the parallel contribution of NH_3 losses to N_2O emissions and particulate matter formation, setting an NH_3 limit value in conjunction with National Emissions Ceilings would offer a win-win-win approach with parallel benefits for habitats, climate and health.

Q4. What is the overall economic cost of nitrogen in the EU environment?

A comprehensive analysis of the costs and benefits of nitrogen in the environment was conducted as part of the European Nitrogen Assessment (Brink et al. 2011; Sutton et al. 2011c). This allowed estimation of the social costs of adverse effects of N_r in the European Environment (see figure 4.3).

The outcomes of the analysis are summarized in Figure 3, which gave a first estimate of the overall cost of European nitrogen pollution at 70 billion to 320 billion Euro per year.

Expressed as € per kg of N_r emission, the highest values are associated with air pollution effects of NO_x on human health (€10–€30 per kg), followed by the effects of N_r loss to water on aquatic ecosystems (€5–€20 per kg) and the effects of NH_3 on human health through particulate matter (€2–€20 per kg). The smallest values are estimated for the effects of nitrates in drinking water on human health (€0–€4 per kg) and the effect of N_2O on human health by depleting stratospheric ozone (€1–€3 per kg).

Of the overall costs, 55-60% is related to air pollution effects on human health. The total damage cost equates to €150–€750 per person, or 1–4% of the average European income and is about twice as high as the present 'Willingness to Pay' (WTP) to control global warming by carbon emissions trading.

The environmental damage related to N_r effects from agriculture in the EU-27 was estimated at €20–€150 billion per year. This can be compared with a direct benefit of N-fertilizer for farmers of €10–€100 billion per year, with considerable uncertainty about long-term N-benefits for crop yield.

Apart from the uncertainties inherent in valuing the environment, including the use of WPT approaches for ecosystem services, the main uncertainties in these estimates concern the relative share of N_r in PM to human health effects and of N_r to freshwater eutrophication effects.

These damage costs are substantially larger than the costs of mitigating NO_x and NH_3 pollution. In particular, a mid scenario considered in preparation for revision of the Gothenburg protocol indicated a cost of €0.6 billion per year, equivalent to 0.8 Euro/kg N abated, highlighting that agriculture is a cost-effect sector for reducing N_r emissions to the environment (Wagner et al. 2011; Sutton et al. 2011c).

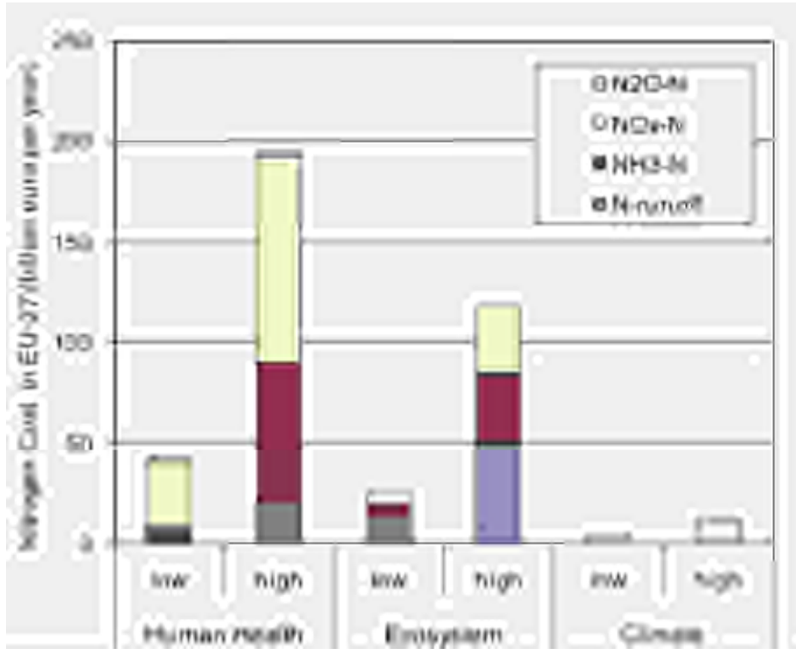

Figure 4.3 Estimated environmental costs due to reactive nitrogen emissions to air and to water in the EU-27 (Brink et al. 2011; Sutton et al. 2011c)

Summary

> The overall costs of nitrogen pollution in the European Union are estimated at €70 billion to €320 billion per year, of which approximately 60% is related to air pollution effects on human health. The total damage of all reactive nitrogen losses to the environment cost equates to €150–€750 per person.

> Environmental damage related to N_r effects from agriculture in the EU-27 was estimated at €20 billion to €150 billion per year.

> Effects of AQ on ecosystems, although harder to value, are estimated to be of comparable order of magnitude to human health effects.

> Revision of NH_3 abatement costs across the EU (mid scenario for the Gothenburg Protocol revision) indicates a cost of €0.6 billion per year, equivalent to 0.8 Euro/kg N abated, highlighting that agriculture is a cost-effect sector for reducing N_r emissions to the environment compared with the social costs.

Q5. How can we improve the valuation of air pollution threats to ecosystems?

One of the ongoing challenges being addressed in the ÉCLAIRE project is how to improve the valuation of air pollution threats to ecosystems, especially considering dose-response relationships for nitrogen and ozone, and the interaction with future climate change.

This ongoing work recognizes that the concept of Ecosystem Services (ES) and research into the quantification of specific services has the potential to close the existing gap of established valuation approaches for ecosystem effects from air pollution. A conceptual framework for this has, for instance, been proposed by Rounsevell *et al.* (2010) and activities within the UK based *Valuing Nature Network* (http://www.valuing-nature.net/) serve as hubs bringing together experts and projects interested in conducting valuation exercises.
One key requirement for an improvement of valuation is the establishment of robust and well documented ecosystem responses to current and future air pollution levels. The work documented in Sutton et al. (2009a), for instance, reflects the wide range of evidence required for the establishment of new ammonia critical levels and loads. Similar to human health effects, the effects of pollutant mixtures requires substantial further research efforts, as these effects are currently not well documented and understood. A second area for improvement addresses the influence of future climate change on the

susceptibility of ecosystems to air pollution effects, respectively their resilience.

Overall it is recognized at present that the valuation of air pollution effects on human health (e.g. Holland et al. 2011) is more advanced than for valuation of effects on ecosystems. Although a first valuation of ecosystem effects has been made in the European Nitrogen Assessment, it is concluded at present that the coupling of dose-response relationships to damage valuation is not yet sufficiently developed for inclusion within Integrated Assessment Models. Further on-going work is therefore needed before the estimated cost of air pollution on ecosystems can be integrated into the optimization of air pollution control measures.

Summary

> The concept of Ecosystem Services (ES) and research into the quantification of specific services has the potential to close the existing gap of established valuation approaches for ecosystem effects from air pollution

> Although a first valuation of ecosystem effects has been made in the European Nitrogen Assessment, the coupling of dose-response relationships to damage valuation is not yet sufficiently developed for inclusion within Integrated Assessment Models, for example to provide a basis to optimise control measures.

Q6. What are the relative costs and benefits of controlling NO_x and NH_3 emissions?

The relative contribution of reduced and oxidised N sources to health and ecosystem effects is a function of both the amounts emitted of these compounds, and the spatial distribution of emissions. In the past 20 years, oxidised nitrogen (NO_x) emissions have been substantially reduced by implementing emission control legislation on road transport sources and large combustion plants (Vestreng et al. 2009). By comparison, there has only been a modest decrease in reduced N (NH_3) emissions during that period, mainly because the agricultural sources have not been subject to the same stringent air pollution controls as already applied to the main NO_x sources. The outcome is that the relative contribution of reduced vs. oxidised N is gradually changing towards a reduced N, as ammonia sources are becoming the dominant emission sources while further NO_2 reductions have been implemented.
The consequence of these changes is that many of the most cost-effective measures for NO_x have now already been adopted, while for NH_3 there are still many cost-effective methods to reduce emissions which have not yet been adopted. In addition, a major review conducted by the Task Force on Reactive Nitrogen under the CLRTAP with the support of the NitroEurope project showed

that NH_3 abatement costs were in many cases much less than previously estimated (UNECE 2011).
These interactions may be visualized in Figure 4.4, which compares the benefit-cost ratio of further controls on NH_3 and NO_x beyond current commitments as of 2011. These graphs incorporate the NH_3 and NO_x mitigation costs as estimated within the GAINS model (e.g. Wagner et al. 2011) and the estimated damage costs of these forms of pollution, based on the European Nitrogen Assessment, as illustrated in Figure 3. While Figure 3 shows that the damage costs of NO_x pollution in Europe are larger than those due to NH_3 (mainly because of the contribution of NO_x to tropospheric ozone formation), Figure 4 shows that it is still substantially more cost effective to mitigate NH_3 emissions. This illustrates how there are still many NH_3 abatement options, which are several times cheaper per kg N abated than for NO_x.

The consequence of these differences is that the cost-effective 'headspace' for further NH_3 emission reduction is estimated to be much larger than for NO_x. Considering the range of uncertainty estimates in Figure 4, there is potential for around 800-1100 kt N further abatement as NH_3 using available technical measures, while only around 100-400 t N using available technical measures for NO_x. Further reduction beyond these amounts would require other non-technical measures, such as would lead to behavioural change in consumption patterns (e.g., vehicle miles and agricultural production).

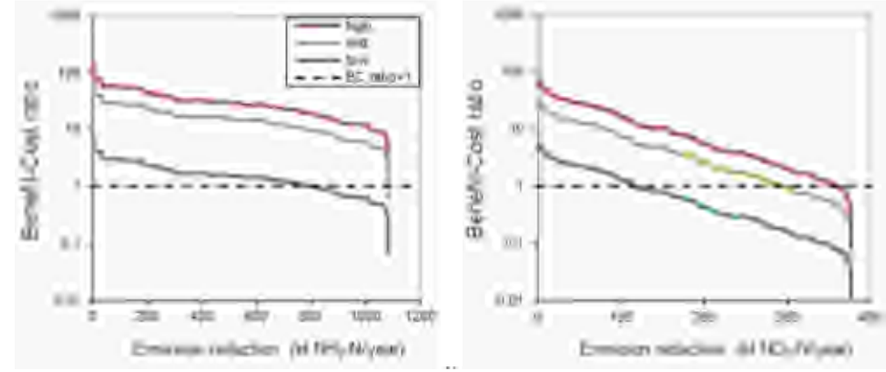

Figure 4.4 Ratio of marginal benefits of emission reduction over the costs of N-mitigation measures in EU27 for NH_3 and for NO_x from stationary sources, for emission reduction from 2010 beyond expected levels in 2020 by effects of current legislation. The comparison shows that there is a much larger potential for cost-effective mitigation of NH_3 emissions (800-1080 kt N) than for NO_x (110-370 kt N) (van Grinsven et al. 2013).

Summary

> With reducing NO_x emissions over the last decades, NH_3 emissions are now a more powerful driver of effects on ecosystems than oxidized nitrogen, making future control of NH_3 emissions a priority.

> As many measures for control on NO_x emissions have already been implemented, further technical measures become increasingly expensive. By contrast, at the European scale, only a few of the available technical measures for NH_3 have so far been implemented, with many low-cost measures still availble. Ammonia therefore offers substantial 'low-hanging fruit' for future air pollution controls.

> Estimated benefit-cost ratios for technical measures (as included in the GAINS model) justify further European reductions in emissions of around 100-400 kt N for NO_x and around 800-1100 kt N for NH_3.

Chapter 5
Air Quality and Climate

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Introduction

The interactions between air quality and climate are complex and include non-linear feedbacks that are not fully understood.

Firstly, human impacts on the atmosphere influence climate forcing either *directly* through the emission of radiatively active greenhouse gases and aerosols, or *indirectly* through the emission of short-lived reactive compounds that influence the atmospheric lifetime and abundance of radiatively active compounds (greenhouse gases and aerosol particles). Figure 5.1 shows an estimate by IPCC (2007) of the radiative forcing by emissions of compounds resulting from human activities during the industrial era. There are large uncertainties not shown here, particularly from the contributions of aerosols.

Secondly, climate change has a significant impact on the chemical composition of the atmosphere and hence on the level of air pollution. Statistical data analyses performed in North America suggest that higher atmospheric temperatures exacerbate photochemical smog and specifically the concentration of surface ozone with a sensitivity factor (called climate penalty factor) of 2.2-3.2 ppbv/K, depending on implemented pollution controls measures (Bloomer et al. 2009). On the basis of such correlation, Lin et al. (2007) suggest that the number of high-ozone episodes in the industrial areas of the north-eastern US could increase by 10-30% in 2030 and double by 2050. Several and sometimes poorly quantified processes contribute to the influence of climate conditions on air quality. Influencing factors include changes in (1) meteorological conditions including the frequency and duration of stagnant air episodes, (2) atmospheric water vapour content, (3) temperature-dependent natural (i.e., biogenic) emissions of ozone and aerosol precursors including volatile organic compounds (VOCs) by trees and nitrogen oxides by soils, (4) emissions of gases and particles by more frequent wild fires, (5) exchange intensity between the stratosphere and troposphere, (6) production of nitrogen oxides by lightning, (7) mobilisation of dust and sea salt particles by wind, (8) intensity and frequency of precipitation and hence of removal processes for soluble species, (9) temperature-dependent formation and destruction rates of reactive com-

pounds and secondary aerosols, (10) dry deposition of species on the canopy and other surfaces, etc..

Many of these processes are best simulated by complex global and regional chemistry-climate or Earth system models. These models account for the two-way relations between the physical climate system, the hydrological system, biospheric processes, biogeochemical cycles, and air quality. Model calculations conducted, for example, by Jacob and Winner (2009) suggest that climate change alone could increase the concentration of surface ozone by 1-10 ppbv over the coming decades, with the largest effects in urban areas during pollution episodes. Changes in the atmospheric abundance of particulate matter (PM2.5) are more difficult to estimate, but could be of the order of 0.1-1 μg m^{-3} in the next decades in response to climate change. Dentener et al. (2005) state that for the "current legislation" case, the annual average ozone levels in the Northern Hemisphere should increase by 5 ppbv from 1990 to 2020. The corresponding higher ozone and methane burdens in the atmosphere increase radiative forcing by approximately 0.2Wm^{-2}. Full application of today's emissions control technologies, however, would bring down ozone below the levels experienced in the 1990s and would reduce the radiative forcing of ozone and methane by approximately 0.1Wm^{-2}. Calculations performed by Katragkou et al. (2011) on the basis of the IPCC A1B scenario suggest that the climate-induced changes in the average European summertime surface ozone concentration between 1995 and 2045 should be below 1 pbbv, with, however, more intense values of 6.2 ppbv in south-western Europe, where temperature-dependent biogenic emissions of ozone precursors will increase considerably as a result of climate change. Several other models have assessed the impacts of air quality control on climate (Menon et al. 2002; 2008; Unger et al. 2009; Jacobson and Streets 2009; Kloster et al. 2010; Zhang et al. 2011).

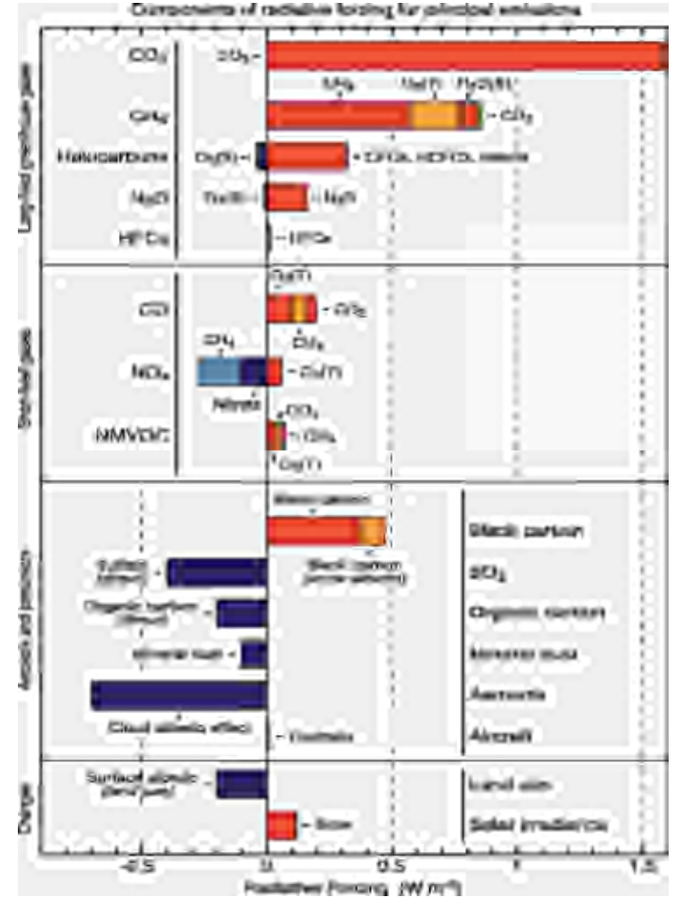

Figure 5.1 Radiative Forcing (RF) for emissions of principal gases, aerosols and aerosol precursors and other changes. Values represent RF in 2005 due to emissions and changes since 1750. (S) and (T) next to gas species represent stratospheric and tropospheric changes, respectively (IPCC 2007.

Q1. What are the synergies between air quality and climate change?

Policies measures for the mitigation of air pollution and climate change overlap, and integrated assessments are required to assess their interlinkages (Colette et al. 2012b). Short-lived climate forcing components (SLCF), such as light absorbing particles (BC), scattering particles (sulphate, OC), ozone and methane, affect both air quality and climate. The reduction of the halocarbon emissions as a result of the Montreal protocol for the protection of the stratospheric ozone layer and its amendments has produced a significant reduction in climate forcing.

Abatement of air pollution is associated with considerable costs, but it can also result in major savings from health improvement and ecosystem effects. However, since major greenhouse gases originate from the same sources as air pollutants, a coordinated abatement strategy is beneficial. Such a strategy should balance reductions of cooling and warming SLCFs and focus on a limited number of sources. It should be made in hand with CO_2 reductions to help achieve climate targets. Technologies, however, can often tackle SLCF more easily than CO_2. The uncertainties are, however, considerable because the indirect effects of aerosols on clouds are not fully understood. Furthermore, the focus until now has been on radiative forcing rather than on the climate response itself (Kulmala et al. 2011).

Balanced reduction strategies of short-lived climate forcers and atmospheric pollutant can enhance the climate mitigation and simultaneously improve air quality (see Figure 5.2). A baseline was developed based on current and planned emission regulation for the period 2000-2030. Three control scenarios adding well known standard reduction measures on BC and methane were investigated together with one scenario where maximum feasible reductions were applied on all substances involved (Shindell et al. 2012).

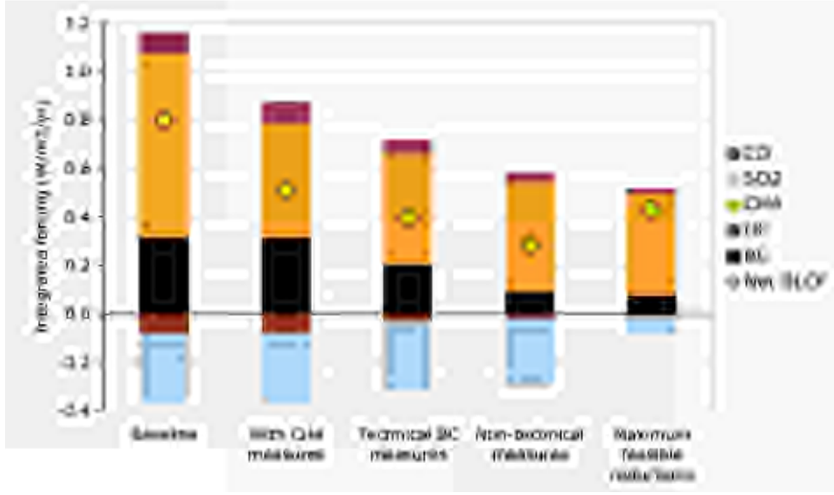

Figure 5.2 Radiative forcing (integrated over 100 years for the baseline, three control scenarios and the MFR on Air Quality regulation, global assessment by substance (UNEP 2011).

Summary

- In many countries, the mitigation policies for climate and air quality are developed by different institutions and ignore therefore the relations between them.
- Many of the tools required to quantify the climate impacts of air pollutants have been developed. Less research has been conducted on the impacts of climate change on air quality.
- Since major greenhouse gases originate from the same sources as air pollutants, a coordinated abatement strategy is the most effective way forward. Such a strategy should prioritize CO_2 mitigation measures that also reduce air pollutant emissions, and complement these measures by balanced reductions of cooling and warming SLCFs.

Q2. Can the climate change and air quality antagonism and synergies be quantified?

If more stringent air pollution abatements were implemented worldwide, the present-day negative total aerosol top-of-the-atmosphere radiative forcing would be strongly reduced in the future (Iversen et al., 2010). Climate change thereafter would therefore be controlled to a larger extent by changes in greenhouse gas emissions. In one particular climate model simulation, the projected temperature response to increasing GHG concentrations and reduced aerosol emissions leads to a global annual mean equilibrium temperature response of 2.2 K (EUCAARI Project 2012). If aerosols were to be abated only in the Industry and Power Plant sector, with the Domestic and Transport sectors staying with currently enforced regulations, the temperature increase estimated by the same model would be 1.9 K. In contrast, a maximum feasible abatement applied only in the domestic and transport sector would lead to a smaller temperature increase of 1.4 K. Enhanced GHG concentrations alone would lead to a temperature response of 1.2 K.

The above results highlight the huge potential impact of future air pollution mitigation strategies on climate and support the need for urgent GHG emission reductions. Since aerosols impact strongly surface forcing and thus have a high hydrological sensitivity, the consequences of plausible precipitation increases associated with global warming would be even stronger. GHG and aerosol forcing is not independent of each other, as they both influence and are influenced by changes in the hydrological cycle.

A climate model study has predicted that, if a maximum feasible reduction (MFR) scenario is applied only to Europe, a substantial warming effect and

increase in precipitation would occur (Kulmala et al. 2011). These effects, however, would be almost double, both over Europe and globally, if the MFR scenario was adopted globally. In the case of current legislation, the temperature increase in Europe was predicted to be 2.2 C, whereas in the case of MFR it would be 4.1 C. The leading driver for this warming effect is the reduction in the SO_2 emissions. These results are based on a single model investigation and their robustness needs to be established by conducting similar simulations with other climate models.

The IPCC RCP emission pathways also prognoses decreases in the sulphur emissions and hence a significant reduction in the aerosol cooling effect. The aerosol concentrations effectively return to pre-industrial levels by 2100 in all RCP scenarios, greatly enhancing the climate warming (Figure 5.3).

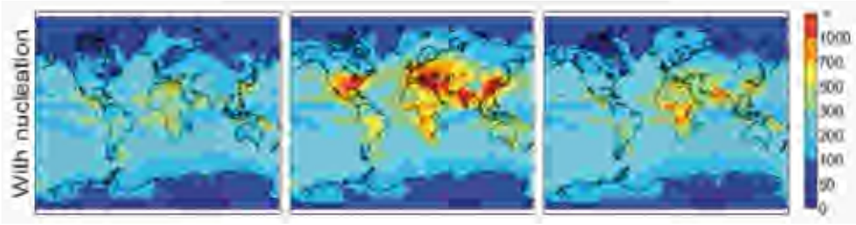

Figure 5.4 Predicted cloud condensation nuclei (CCN) concentrations in the preindustrial (left), current day (middle) and 2100 (right) conditions. These changes in CCN are proportional to the aerosol cooling effect, effectively reducing the aerosol forcing back to pre-industrial levels. This change is mostly attributed to reduction in SO_2 emissions predicted in the IPCC RCP emission pathways (Makkonen et al. 2012).

Removing black carbon does not have similar side-effects on climate as removing sulphur emissions. In 2010, UNEP performed an assessment of BC and ozone that was recently published using the same scenarios (www.unep.org/dewa/). The evaluation by two GCMs found that the global temperature increase could be reduced by about 0.4°C from 2030 and onwards with the suggested methane and BC control measures, and at the same time considerable reductions in adverse health and ecosystem effects were achieved.

Changes in ozone precursor gases generally have smaller effects on climate than aerosols. The exception to this is methane, emissions of which since 1850 have resulted in a forcing (direct and indirect) of 0.8-0.9 Wm^{-2} (Figure 1). Future emissions of methane are expected to increase in the highest scenario (RCP 8.5) but to decline by the end of the century in other scenarios. NO_x, CO and VOC emissions are expected to decrease after about 2030 (Figure 5.4). All generate tropospheric ozone and so increase its radiative forcing. CO and VOCs also decrease the atmospheric oxidising capacity and hence increase the lifetime of methane and adding additional radiative forcing to the atmosphere. Conversely NO_x emissions decrease the methane lifetime, which leads to a

negative forcing. Combing the ozone and methane effects, NO_x emissions lead to a slight net decrease in radiative forcing (although consistent with zero) as shown by the HTAP multi-model experiment (Fry et al. 2012 and Figure 5.5). These factors imply that methane, CO and VOC reductions lead to win-win situations, improving both air quality and climate. NO_x emission reductions lead to win-lose situations benefitting air quality by warming climate. However in many cases NO_x control measures are the most effective at improving ozone, so the small warming may be acceptable.

Mitigation of CO_2 emissions will affect co-emitted species. In many cases, reduced combustion of fossil fuels will lead to lower SO_2, NO_x and PM emissions and thereby improve air quality as a co-benefit of the CO_2 reduction. However, climate strategies that enhance the use of biomass could increase emissions of SLCF, particularly of fine particulate matter, with negative impacts on human health. Reductions of methane emissions will benefit climate not only directly but also through reduced formation of tropospheric ozone

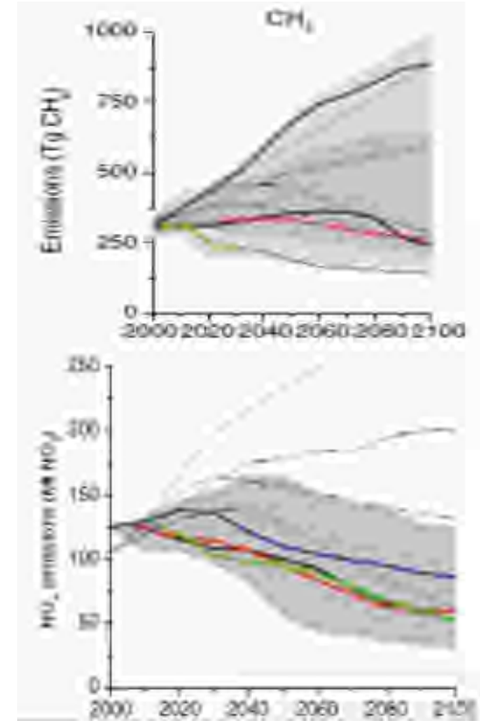

Figure 5.4 Methane and NO_x emissions in the RCP scenarios.

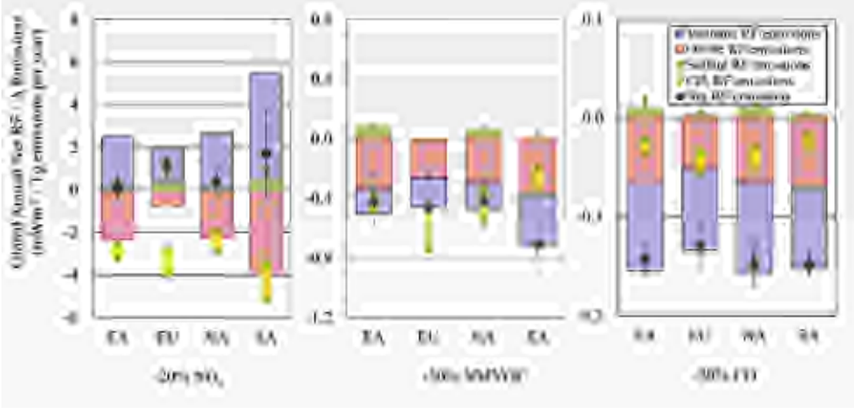

Figure 5.5 Radiative forcing from NO_x, VOC and CO emissions. (from Fry et al. 2012).

Air quality pollutants can also affect climate via the biosphere. For instance ozone damages vegetation, whereas nitrogen deposition acts as a fertiliser. By diffusing the incoming solar radiation, aerosols can increase the efficiency of plant photosynthesis (Mercado et al. 2009). These biospheric feedbacks are not negligible and lead to composition changes on a much longer timescale that the lifetimes of the air quality pollutant (Mahowald 2011). Collins et al. (2010) found that the reduction in carbon sequestration due to ground level ozone could be enough to turn NO_x emissions from net cooling to net warming.

Summary

- Great potential exists for air pollution mitigation strategies to supplement climate policies to limit temperature increase in the near term.
- Removing black carbon and reducing methane would simultaneously benefit air quality and climate in the near time.
- If stringent air pollution abatements are implemented worldwide, the present-day negative total aerosol top-of-the-atmosphere radiative forcing will be strongly reduced (perhaps by 50 %) by 2030.
- IPCC RCP assumptions effectively return aerosols to pre-industrial levels by 2100 in all scenarios, greatly enhancing the climate warming; however, embedded in the RCP emission scenarios is the assumption that stringent air quality policies will be adopted, which will not necessarily happen without further policy interventions.

The assumption made in the RCP scenarios will become reality only if explicit policies are developed to deliver the projected reductions.

Q3. What is the effect of climate policy scenarios on achievement of air quality objectives?

Reductions of CO_2 emissions as a consequence of air quality policy measures will be beneficial for climate, especially in the long run. However, many ("end-of-pipe") air pollution control measures will not affect CO_2 emissions, or will slightly increase them.

Mitigation of CH_4 (and CO) emissions would reduce hemispheric background ozone levels and provide near-term climate benefits, both through lower CH_4 and ozone concentrations. Such measures could complement conventional local and regional ozone strategies focusing on NO_x and VOC, which have much lower net climate impacts.

Combustion emits a mixture of cooling (e.g., sulphur) and likely warming (e.g., black carbon) aerosol components or their precursors, as well as precursors (NO_x, CO and VOC) of the greenhouse gas ozone. The mix between the various components is source-specific. Air quality strategies considering also their climate impacts should give preference to measures that reduce the warming components.

Current legislation will lead to further reductions of sulphur dioxide emissions in Europe. Since sulphate aerosols produce a strongly cooling, this reduction in aerosols will likely lead to extra warming during the next few decades. Faster introduction of cleaner non-road mobile machinery as well as retrofitting existing vehicles could minimize this additional warming through lower emissions of black carbon (BC)-rich sources. The ban of burning of agricultural waste and the reduction of emissions from the domestic use of solid biofuels would also help to reduce BC emissions, but as these sources also emit large amounts of organic carbon (OC) their net effect is less certain.

Air quality-driven reductions in global anthropogenic SO_2 emissions are likely to decrease the cooling effect of aerosols during the next hundred years due to the impact of SO_2 on aerosol mass concentrations and cloud droplet number concentrations via secondary aerosol formation. This effect is likely to overwhelm the potential changes in natural emissions of aerosol precursors.

Different models have been used to simulate the response of the climate system to air quality improvement. For example, Brasseur and Roeckner (2005) have shown that removing the totality of anthropogenic sulphate aerosols from the atmosphere would produce a rapid mean warming of the planet of the order of 0.8 K, with the most pronounced effects to be expected in Polar Regions. More recent model predictions performed by the EUCAARI project have shown that, halving the SO_2 and anthropogenic primary particle emissions would result in reductions of the order of 20% in the total particle number concentrations. This suggests that natural aerosol production might somewhacompensate for

the reductions in the anthropogenic primary aerosol emissions (Kulmala et al. 2011).

Emission scenarios investigated with the GAINS model were found to significantly affect the climate of the whole of the Northern Hemisphere (Kulmala et al. 2011). The different member states' contributions to the climate effect in Europe vary to some extent with their geographical location. The Arctic rim countries have a larger effect on the Arctic compared to the European countries further south. Climate change affects air quality, but it was found that air quality mitigation scenarios will give a clearly significantly better and noticeable improvement in air quality in spite of the variations induced by climate change and natural variability in the climate.

Summary

- Reduction in the emissions of methane, and of absorbing aerosols, in particular black carbon, will contribute to an improvement of air quality, specifically to a reduction in the ozone concentration. Reduction of SO_2 emissions, however, should exacerbate global warming in the near term.
- Climate change affects air quality, but air quality mitigation measures will be more effective in improving air quality in spite of the variations induced by climate change and natural variability in the climate.

Q4. Will the effects of Climate Change this century increase or decrease emissions of air pollutants in Europe

Natural emissions of several gases exhibit strong temperature dependence. This is the case of biogenic volatile organic compounds that are precursors to secondary pollutants such as ozone and organic aerosols. The emission of isoprene (C_5H_8), essentially by deciduous trees, increases considerably with temperature. It is also affected by other factors such as light intensity, climate conditions including the atmospheric level of CO_2 (Guenther et al. 1993; Arneth et al. 2007). In the presence of nitrogen oxides and light, the chemical degradation of isoprene represents of major source of tropospheric ozone. Emissions of methane, a greenhouse gas and another precursor of ozone, by anaerobic processes in wetlands are also temperature dependent. Climate change is expected to produce more frequent wildfires with consequently increased emissions of carbon monoxide, VOCs, nitrogen oxides and black carbon.
A large mass fraction of the European aerosol is organic, and a large fraction of that aerosol is made of modern carbon (i.e. deriving from non-fossil fuel sources). The main sources for modern carbon in Europe include wood combustion and secondary biogenic organic aerosol (BSOA). Since the strengths of these two sources are expected to vary in response to climate change, modern carbon might be a part of an important feedback mechanism in the climate system. Climate warming of a few degrees leading to increasing monoterpene emissions will enhance future BSOA formation. Monoterpene emissions of Mediterranean tree species depend stronger on temperature, leading to stronger BSOA formation over the Mediterranean compared with Boreal regions for the same degree of warming. Simulations based on current knowledge of BVOC response to temperature increases show increases in BVOC emissions with temperature followed by surface ozone concentration increases (~1 ppbv/°C in the East Mediterranean) and smaller increases in PM due to partial compensation in changes of sulphate by those in OC (Im et al. 2011). These results suggest that the biogenic organic aerosol content in the European aerosol will increase with the climate warming, and could decrease the effectiveness of air quality measures in the future. However, since the domestic heating will probably decrease with increasing temperatures, the overall effect of climate change to aerosol concentrations is unclear.

Summary

- Natural emissions of several pollutant gases by the ecosystems and soils (i.e., ozone and aerosol precursors) exhibit strong dependence on climate (i.e., temperature and hydrology). Climate change will generally increase these natural emissions
- Climate change could therefore erode the benefits of air quality policies.
- Anthropogenic emissions of greenhouse gases such as CO_2 could also change in response to climate change.

Q5. Will the effects of climate change this century increase or decrease the lifetimes and transport distances of air pollutants in Europe?

Under a warmer climate, the atmospheric abundance of water vapour is expected to increase significantly. As a result, the formation rate of the hydroxyl (OH) radical will be enhanced. Since OH is the major atmospheric oxidant that controls the *in situ* destruction of many pollutants, climate change could lead to a reduction of the lifetime of several radiatively active gases, and hence a reduction in the transport distances of air pollutants. However, the magnitude of these effects is relatively small at least on global scales. Models predict, for example, a climate-generated reduction of ozone in most of the free troposphere, in response to increasing concentrations of water vapour. An increase in the abundance of OH should also slow down the trend in methane expected from increasing emissions associated with more intense agricultural activities

and with increasing temperature-dependent emissions by wetlands. The issue, however, is complicated by the fact that the OH radical is destroyed by reaction with carbon monoxide and produced by reaction by nitric oxide. The atmospheric concentrations of these two compounds are expected to change in the future as a result of air quality improvement, with impacts on the atmospheric OH level.

Aerosol lifetime is determined primary by the mean size of the particle population (dry removal of particles), its water-solubility (both dry and wet removal), and the frequency and rate of precipitation (wet removal). An increase in ambient temperature is likely to increase secondary aerosol formation from biogenic sources. This, together reduced sulphur emissions, is likely to decrease the mean size of submicron aerosol, as well as their solubility. Plausible increases in precipitation would enhance aerosol wet removal rates in all size ranges. The overall effect of these changes in aerosol lifetime in complex and difficult to estimate based on current knowledge. It is very likely, however, that changes in aerosol lifetimes vary from region to region in Europe.

Current climate models do not agree on the sign of the change for aerosols. Changes in clouds with climate warming drive aerosol changes, and there is model diversity in those cloud responses. Racherla and Adams (2006) and Liao et al. (2009), both using the GISS model, obtain a decrease in aerosol residence times as wet deposition increases. But Bellouin et al. (2011), using the Hadley Centre climate model, obtain the opposite effect as they find that aerosols are less likely to meet low and middle clouds.

Summary

- With the expected increasing concentrations of water vapour in the atmosphere, models predict a climate-generated increase in the concentration of the OH radical and hence a reduction in the atmospheric lifetime of greenhouse gases such as methane and ozone in most of the free troposphere.
- The impact of climate change on aerosol concentration and removal rate is not well quantified. Even the sign in the change for aerosols is uncertain.
- Changes in climate-generated meteorology (e.g., air stagnation episodes, cross tropopause exchanges) could affect the atmospheric lifetime and concentrations of primary and secondary pollutants, exacerbating uncertainties in future air quality.

Q6. Are there regions particularly sensitive to European pollutant emissions?

The temperature response to regionally distributed radiative forcing occurs at broader scales than the radiative forcing itself, due to the transport of generated heat in the atmosphere. The scales of these temperature responses are intercontinental, and reliable quantification of the response by current climate models is probably possible only in broad zonal bands.

Transport of pollutants from Europe to the Arctic, for example, is relatively efficient especially in the lower troposphere, and European emissions therefore have a strong impact on the near-surface concentrations of short-lived climate forcers in the Arctic (Quinn et al. 2011). The deposition of black carbon on high-latitude snow and ice surfaces produces a positive radiative forcing which is believed to have a high efficacy, which is likely enhancing spring melt and contributing to Arctic warming.

Chapter 6
Integrated Assessment

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Introduction

There are important interactions and potentially large economic synergies between air quality strategies and the objectives of EU social and economic policies, including the EU policies on energy, agriculture, transport and climate change. These complex - and well documented - interactions impose formidable challenges to decision makers. Unless incorporated into the process, these interactions could prohibit cost-effective solutions and also unnecessarily waste important resources of Europe's economy.

Integrated assessment, based on latest scientific findings and validated data, can provide valuable information on the design of cost-effective strategies that meet multiple policy objectives. Integrated environmental assessment brings together knowledge across scientific disciplines with the purpose of providing relevant information to decision makers.

In the past, integrated assessment modelling has been extensively used as an analytical backbone of the Clean Air For Europe (CAFE) programme of the European Commission that provided the knowledge base for the Thematic Strategy on Air Pollution (TSAP).

Since then, methodologies for integrated assessment modelling have been further developed at the European level along multiple avenues. Numerous EU-FP6 and FP7 projects addressed specific aspects that are relevant for air pollution control strategies. Under the EC4MACS project of the EU-LIFE program, a toolbox of well established modelling tools that have already been used for earlier policy analyses has been developed to explore the synergies and interactions between climate change, air quality and other policy objectives.

The EC Framework Programmes and other directed research contribute to the methodological development as well as underpinning scenario and policy analysis

Q1. **How can policies be developed to address impacts of both air pollution and climate change?**

Policies that reduce air pollution, in most cases also influence greenhouse gas emissions and policies that are designed to reduce greenhouse gas emissions in most cases also decrease or increase emissions of air pollutants. A recent paper in Nature by Shindell et al. (2012) illustrates the synergies between climate change (specifically, the near-term effects of reducing ozone and black carbon, both short-lived climate pollutants, SLCPs), human health and food and energy security. The GAINS model has been extended to consider the impacts of ~3,000 specific mitigation measures on the emissions of short- and long-lived substances, and to quantify their impacts on human health, ecosystems, and radiative forcing. With appropriate quantification, in particular of the climate impacts, the GAINS optimization provides a methodology to develop cost-effective portfolios of measures that maximize the benefits on human health, vegetation and climate change.

The FP7 Climate Cost project assessed the co-benefits of global post-2012 greenhouse gas mitigation strategies on air pollution (Rafaj et al. 2012). It was found that in 2050 expenditures on air pollution control under a global climate mitigation regime would be 250 billion € lower than in a business-as-usual scenario. Around one third of financial co-benefits estimated world-wide in this study by 2050 occur in China, while an annual cost saving of 35 billion € is estimated for the EU if the current air pollution legislation and climate policies are adopted in parallel. Health impacts of air pollution are quantified in terms of loss of life expectancy related to the exposure from anthropogenic emissions of fine particles, as well as in terms of premature mortality due to ground-level ozone.

Related activities have examined links between the nitrogen cycle and climate change (NitroEurope) and a report compiled under UNECE Task Force on Reactive Nitrogen has specifically investigated the interactions between nitrogen and climate change (http://www.nitrogenweb.info/). A case study assessing climate change policies carried out in the projects INTARESE and HEIMTSA (2011a) indicates that the effects of climate policies on reducing air pollution and human health impacts, when expressed as external costs, are of a similar magnitude as the effects these policies have on mitigating climate change. While climate policies measures often reduce human health impacts, some climate measures may increase health risks, e.g., increased biomass burning in small stoves leads to higher PM concentrations, or improved insulation of buildings with new air-tight windows increasing mould formation.

The concept of ecosystem services (ES, e.g. Compton et al. 2011), may provide an as yet missing link to quantify (and methods for the valuation of) ecosystem effects that have proven elusive in the past, often leading to a bias towards the

assessment of health effects in cost-benefit analyses. Other approaches for the assessment of biodiversity losses have been developed in NEEDS and in the 'European Nitrogen Assessment' (Sutton 2011b).

The key challenges for a full integration of the assessment of climate change, human health and ecosystem damage lie in the substantial difference in spatial and temporal aspects. A modular model system with well-defined interfaces, which allow exchanging some elements (models) with other elements depending on the question addressed – a concept developed in HEIMTSA (2011b) provides a possible way forward.

Summary

- Policies to reduce air pollution in most cases also influence greenhouse gas emissions and policies that are designed to reduce greenhouse gas emissions in most cases also influence (decrease or increase) emissions of air pollutants. Furthermore some air pollutants are short lived GHGs, and climate change affects emissions of pollutants, their transport and the chemical transformation in the atmosphere. Thus an integrated assessment should assess impacts of air pollution (on human health, biodiversity, crop yield) and impacts on climate.
- To assess biodiversity losses or gains in monetary terms, some approaches (e.g. in the context of ecosystem services) have been developed that should be tested for their usefulness in future policy analyses.
- It has been suggested to employ monetized benefits as a common metric to compare impacts of air pollution and climate policies. However, such monetizations are loaded with complexities, related, e.g., to the robust quantification of the value of human life, the monetization of ecosystems impacts, and the comparison of benefits that occur at different temporal and spatial scales.
- An integrated assessment tool box should consist of modules that are linked via well-defined interfaces. The tools to be used are then chosen depending on the questions to be answered. The central provision of data needed like meteorological data, emission data, population data would help to support the assessment.

Q2. Are the scales of assessment for the different pollutants (from local scales of urban air quality to the global scale assessment of radiative forcing by GHG) properly considered in IA?

The challenges presented by conducting integrated assessment across a range of scales have been discussed by Reis et al. (2012) and previously by Oxley and ApSimon (2007). While it is tempting to aim for a 'one-size-fits-all' approach, models should only be applied for the spatial scales they have not been designed. Approaches like nesting local and national scale IAMs within more coarse European-wide models is not sufficient to make a localized analysis covering the whole of Europe, which however is necessary to assess EU policies. Thus, for the integration of local scale effects into European wide IAMs, a parameterisation of local scale effects could be used to derive an "urban increment", which is described in the following section. In addition, networking and knowledge exchange of groups developing and applying IAMs at different scales is beneficial for the methodological development of IA concepts, e.g. through the Network of Integrated Assessment Modellers (NIAM, www.niam.scarp.se).

Estimation of the urban increment: In the Megapoli project, a Eulerian model or a parameterised version of a Eulerian model is used at the European scale to estimate average annual concentrations (and deposition) of primary and secondary pollutants for each grid cell of the model domain. A grid element usually has a size of between 50km*50km and ca. 7km*7km for European wide calculations. In such a grid, there could be as well cities or city parts as non-habituated areas. From measurements it is known, that PMx background concentrations are higher in cities than outside cities. As humans are staying within the city boundaries most of the time, the concentration in the city (and not the concentration in the grid) should be used to estimate health impacts. A first attempt to generate a method to estimate the 'urban increment' was made within the 'City Delta' project (http://aqm.jrc.it/citydelta/). This attempt was considerably improved within the project MEGAPOLI (Moussiopoulos et al. 2011; Torras Ortiz 2011). Especially the use of accurate emission data for a city is an important input parameter for estimating the urban increment.

Estimation of the distribution of concentrations in cities: Measurement stations, where EU air quality limits for NO_2 and PM10 are exceeded, are often located in street canyons; furthermore people's homes are located near streets. So to analyse, whether thresholds are met and to get a better picture of the exposure of people to pollutants, not only the background concentration, but in addition the distribution of street canyon concentrations should be known. To model street canyon concentration in single streets or in a whole city, a number of models are available, e.g. those further developed and improved in TRANSPHORM. However, an application of these models for all cities in Europe in the frame of a EU wide assessment is not feasible. Instead, a statistical approach developed in MEGAPOLI could be used (Torras Ortiz 2011). However this model is only validated for German cities, thus some further work is needed, until it can be applied for all European cities.

Estimation of emission data: The scale problem also exists with regard to emission data. Annual line and area sources are first estimated per country and then distributed spatially using proxy data and spatially using temporal patterns. Recent analyses in TRANSPHORM revealed that part of discrepancies between modelled and measured PM10 concentrations might be caused by a poor temporal and spatial distribution of emission data. This could be improved by taken current improvement in the methodology for the temporal and spatial resolution into account, e.g. that emissions depend on temperature (Theloke et al. 2011; Theloke et al. 2012; Thiruchittampalam et al. 2012; Vogel et al. 2012) and on-going work e.g. in the MACC project (http://www.gmes-atmosphere.eu/) and EDGAR (http://edgar.jrc.ec.europa.eu). However, work for the EC4MACS project also highlighted clear limitations of generic downscaling processes for emission inventories, which constrain the improvements that could theoretically gained from a more spatially resolved modelling of air quality.

Summary

- Owing to the EU governance system and the transboundary transport of pollutants in Europe, air pollution policy assessments need to be conducted for the entire EU, while exposure and some measures are very local in nature. This poses challenges that have been addressed in recent FP projects. Especially for the estimation of the 'urban increment', i.e. the difference between the urban background concentration and the average concentration in the grid(s) around the city, and the estimation of the distribution of the 'street canyon increment' methods have been generated or improved.
- Methods for a better spatial and temporal resolution of emission data have been developed and should be used to improve the quality of spatially and temporally resolved emission data.
- For future air quality reviews additional insights could be gained by adding exposure modelling to the assessment.

Q3. Should the assessment of non-technical measures be included in an integrated assessment?

Cost-benefit analyses for environmental policies usually only assess costs and impacts of technical measures, i.e. measures that change the emission factor of a process, e.g. by adding a particulate filter or by increasing the effectiveness of the filter. However, as many of the technical measures are already requested by current legislation the scope for further improvements of end-of-pipe technologies is often limited.

In such cases, further substantial reductions of emissions are only possible, if non-technical measures are used; in many cases such measures require behavioural changes of people. For instance, people might refrain from using less environmentally friendly processes resp. technologies or substitute them with other better processes. This could be achieved by regulating the use of the unwanted process or by increasing the price of the unwanted process or by subsidizing the use of other more environmentally friendly processes. For example, if the tax on gasoline and diesel is increased, some users of private cars might shift to riding a bicycle or using public transport. In the projects MEGAPOLI and INTARESE it was shown, that an effective and efficient non-technical measure for reducing air pollution as well as climate change is a change of diet, especially a reduction of the consumption of meat. However, as pointed out in EC4MACS, these impacts would be limited unless milk production would be reduced simultaneously, as the current agricultural production system in Europe is closely coupled.

The costs of such measures should include the utility losses experienced by those that change their behaviour. In the example above, private car users shifting to public transport might even save money, but before the tax increase they have been willing to use their car despite the higher costs, as they feel they experience some utility gains by using their car like more comfort or better time management and no waiting times.

Thus the analysis of non-technical measures would be an important part of an integrated assessment. Examples can be found in results for the FP7 project MEGAPOLI (megapoli.info) and the German project 'PAREST' (Appelhans 2012).

Summary

- As the potential for further emission reduction from technical measures, i.e. measures that reduce emission per unit of activity, is decreasing with the stringency of emission control legislation, policy assessments should also include non-technical measures, i.e. measures that change the behaviour resp. the decisions of people. In addition, even when assessing technical measures, changes in behaviour should be considered.
- The European Commission identified the field of integrated assessment as one of the priority issues for the review and implementation of the air quality policy and has recently funded a new coordination action within FP7 (APPRAISAL). The goal is to provide scientific and technical support in this area to both the European Commission and EU Member States and regions during the on-going revision and the implementation of the EU Air policy.

Chapter 7
Future research needs

In this chapter are reported, for each topic, the main priorities in future research needs that have been identified during thia process.

The list, even if far from being exhaustive, is nevertheless intended to give an indication on where some of the future research efforts should be directed.

Air quality and health:

In order to better understand how air quality could affect human health, it would be important to quantify the role of the different components of PM, as well as to improve the spatial and temporal resolution of PM distribution.

More studies are needed in order to estimate the total PM exposure of individuals, e.g. from ambient, occupation, active smoking.

European studies on long-term effects of elevated background O_3 concentrations are lacking. However, it would be extremely important to separate and quantify the effect of long-term exposure to O_3 as well as to quantify the burden of disease due to O_3.

Future studies should include aggregate indices of exposure to shed more light on the usefulness of a single health related objective.

As far as NO_2 is concerned, causal link between exposure to NO_2 and health effects must be investigated.

Particulate matter

Spatial and temporal representativeness of emission inventories is often limited and the consistency across scales needs improvement. Efforts are needed to improve emission inventory completeness and to include species so far ignored, such as intermediate volatility organic compounds (IVOCs), which might be significant source of secondary organic aerosol at local, regional, and global scale.

Policy measures to improve air quality need support from air quality models. Several models have limitations in predicting secondary aerosols, especially secondary organic aerosol. New model approaches, that include intermediate volatility organic compounds (IVOCs), better describe organic aerosol loadings, but organic vertical distribution and temporal variability is poorly described.

A few laboratory studies highlight a correlation of mortality and morbidity in hot-spot urban areas with ultrafine particle (UFP) concentration and composition (e.g. Verma et al. 2011). UFP (particles with aerodynamic diameter smaller than 100 nm) represent less than 10% of fine aerosol mass in most urban locations, but comprise more than 90% of particle number concentration. UFP in urban atmosphere are mainly emitted by vehicular road traffic, through incomplete combustion of petrol and diesel fuel. Although the current European background levels of UFP have been characterized (Asmi et al., 2011), there is still not enough information on particle number health effects to suggest whether an ambient number concentration limit value should be considered.. Continuous monitoring of UFP could be suggested as an optional improvement of present monitoring activity in view of future revisions of AQ legislation.

The majority of toxicological and epidemiological studies focus on a single fraction of PM, generally PM10 or PM2.5, although it is reasonable to expect a dependence of heath effect on particle size, composition, and mixing state, which depend on particle source and processes. Future revisions of air quality directive require a better knowledge of the health effects and action mechanisms of the different PM sources, focusing also on different size fractions, i.e. UFP, PM1, PM1-2.5, PM2.5-10.

Although epidemiological studies show a correlation between BC and health effects, toxicological investigations are needed to understand the role of BC as active toxic species or carrier of other agents, such as transition metals and/or aromatic organic species.

Ozone

While the broad scale features of tropospheric ozone trends over the last century are reasonably well captured by global CTMs application to quantify effects at the surface remain constrained by their coarse resolution.

The simulation of long term trends in surface ozone at measurement sites throughout Europe remains unsatisfactory

Nitrogen

Improvements in agricultural efficiencies throughout the world and perhaps also changes in dietary habits in the developed world are required to reduce losses of reactive nitrogen to air water and the oceans. Research to identify the most effective strategies is required.

The need to produce more food as human populations grow and as nutrition improves in the developing world will make N_2O mitigation especially challenging. The AR5 RCPs for N_2O properly express the large range of possibilities, from continuously increasing atmospheric N_2O at present rates under RCP8.5 with little improvement of agricultural efficiencies, to declining N_2O concentrations later this century under RCP3PD. The latter possibility would require several highly effective mitigation efforts in all sectors as well as reduced per

capita meat consumption in the developed world. Improvement of Resource use efficiency is not relevant to reduce N_2O emissions, but is the key to mitigating all N emissions.

Great progress has been made in NO_x mitigation in the developed world, although further improvements are needed and possible. NO_x mitigation is urgently needed in many developing world countries, especially emerging market countries. Due to the interactions of temperature, NO_x concentration, and VOC concentration, the effectiveness of NO_x mitigation to reduce harmful ozone and particulate matter concentrations is climate-sensitive, making effective NO_x mitigation for ozone reduction more difficult in a warmer world. The effectiveness of carbon sequestration in forests and soils will depend upon C-N interactions. Estimates of carbon sequestration at present and in the future will be inaccurate unless these interactions are properly accounted for.

The atmospheric processing of NO_y and NH_x compounds remains unsatisfactory to explain current trends and reliably quantify interactions with the biosphere and clouds.

The below ground N cycle is poorly modelled, while at the same time models are very sensitive to how N cycling is simulated.

The CO_2 fertilization effect is likely overestimated by models that do not include reactive N (N_r) and progressive N limitation is likely to cause a diminished CO_2 fertilization effect in the future. Including N in carbon models will, therefore, result in higher projected CO_2 concentrations in 2100, compared to using carbon models alone.

We know that global biogeochemistry models get incorrect answers if they do not include N, but our skill at modelling C-N interactions at global scales is still poor. This also holds for the modelling of the relationship and feedbacks from the phosphorus cycle and hydrology.

Understanding the effects of Nr on ecosystems has some way to go to quantify responses of ecosystem services to Nr deposition (exposure) and remains a research substantial challenge.

Air Quality and Climate

We need to better understand how climate change, and mitigation strategies, will affect future air quality. This is needed to avoid that climate change could make ineffective air quality measures, and to design climate change mitigation strategies that maximize co-benefit for both air quality and climate.

Integrated Assessment

The power of current and future IAMs undoubtedly is that they present a comprehensive conceptual model of all known or quantifiable relationships between the drivers of change and their resulting effects.

By establishing integrated assessment as a paradigm for ex-ante (as well as ex-post) policy assessments, the likelihood of unintended consequences of policy measures leading to a reduction of efficiency or, in the worst case, a complete offset of intended consequences can be significantly reduced.

To achieve this, it is vital that IAMs are able to incorporate the latest scientific findings, while at the same time providing robust and consistent answers to policy questions. This, in itself, may present the greatest challenge in a dynamic and fast changing policy field.

Air Quality Monitoring

The future challenges for urban air quality monitoring in Europe are to fill the gaps in information related to spatial and temporal variations of exposure to health-relevant air pollutant metrics. Routine monitoring should address multiple purposes e.g. compliance assessment, effectiveness of AQ action plans, routine health monitoring and assessment, and impact assessment. Therefore following R&D steps are needed:

Areas for Research and Monitoring of Air Quality (ARMAQ,) focussed on human health have to be developed in various densely populated areas in Europe, to enable the development and integration of new measurement technologies, measurement strategies, data integration and analysis tools, as well as testing new exposure metrics (e.g. EC) and exposure assessments.

These ARMAQs should be closely linked to health effect studies by including routine health data recording, health effect studies, and health impact assessments, including using different types of cohorts. This may be extended in a second step to the inclusion of other environmental stressors to allow more complete health effect studies.

Black carbon (calibrated vs. elemental carbon, EC) is one promising additional air quality parameter to be regulated in revised AQ standards, since it has been demonstrated that there is a high cause-effect relationship with health outcomes. It is also relevant that, aside from the health aspect, high time resolution instruments are available to monitor BC, and this is a good indicator of the impact of road traffic on air quality in urban areas. Black Carbon / Elemental carbon could therefore be one of the first parameters tested in the framework of the first bullet points.

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European Commission

Research Findings in support of the EU Air Quality Review

Luxembourg: Publications Office of the European Union

2013 — Number of pages pp.108 — 16,5 x 23,5 cm

ISBN 978-92-79-29457-0

doi 10.2777/98974

ki-30-13-524-EN-C

Presented during the EU Green Week in June 2013, the report «Research findings in support of the EU Air policy review» syntheses the output of the main projects funded by the EU framework programme in areas closely related to the field of air quality. It contributes to the «Science and Policy interface» activity promoted by DG RTD consisting in the launch of a major coordinated effort with the scientific community to address the specific research needs for the Implementation and review of ambient air quality, the National Emission Ceilings (NEC) Directive and the Thematic Strategy on Air Pollution.

Research and Innovation policy

Publications Office

ISBN 978-92-79-29457-0

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