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RAINS REVIEW 2004

The RAINS model.

Documentation of the model approach prepared for the RAINS peer review 2004

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The Regional Air Pollution Information and Simulation (RAINS) model

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Purpose of this document

This documentation provides background information to the reviewers of the RAINS 2004 peer review, conducted for the Commission of the European Communities, Directorate General for Environment, by a review team led by Prof. Peringe Grennfelt.

Acknowledgements

The RAINS model brings together work from a large group of scientists from a wide range of disciplines. Thus, the development of the RAINS model would not have been possible without the constructive cooperation of so many colleagues.

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1 General approach

1.1 The RAINS model

The Regional Air Pollution Information and Simulation (RAINS) model developed by the International Institute for Applied Systems Analysis (IIASA) combines information on economic and energy development, emission control potentials and costs, atmospheric dispersion characteristics and environmental sensitivities towards air pollution (Schöpp *et al.*, 1999). The model addresses threats to human health posed by fine particulates and ground-level ozone as well as risk of ecosystems damage from acidification, excess nitrogen deposition (eutrophication) and exposure to elevated ambient levels of ozone. These air pollution related problems are considered in a multi-pollutant context (Figure 1.1), quantifying the contributions of sulphur dioxide (SO_2), nitrogen oxides (NO_x), ammonia (NH_3), non-methane volatile organic compounds (VOC), and primary emissions of fine (PM2.5) and coarse (PM10-PM2.5) particles (Table 1.1). The RAINS model also includes estimates of emissions of relevant greenhouse gases such as carbon dioxide (CO_2) and nitrous oxide (N_2O). Work is progressing to include methane (CH_4) as another direct greenhouse gas as well as carbon monoxide (CO) and black carbon (BC) into the model framework.

Table 1.1: Multi-pollutant/multi-effect approach of the RAINS model

	Primary PM	SO_2	NO_x	VOC	NH_3
Health impacts:					
- PM	√	√	√	√	√
- O_3			√	√	
Vegetation impacts:					
- O_3			√	√	
- Acidification		√	√		√
- Eutrophication			√		√

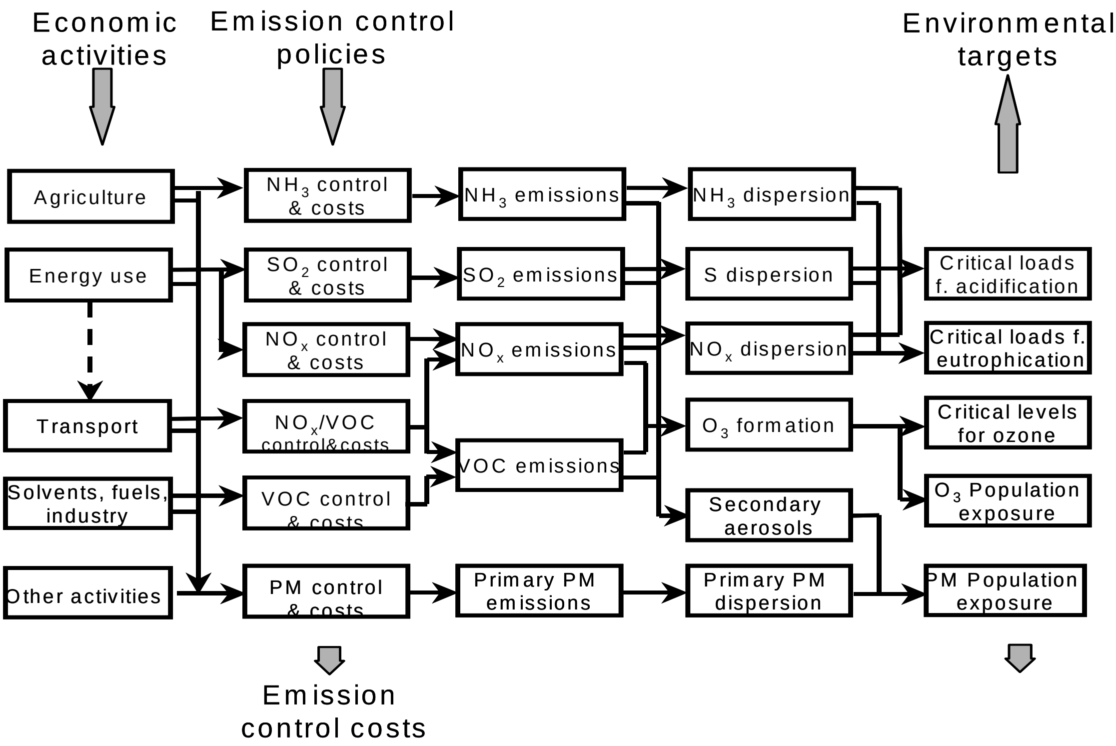

Figure 1.1: Flow of information in the RAINS model

A detailed description of the RAINS model, on-line access to certain model parts as well as all input data to the model can be found on the Internet (http://www.iiasa.ac.at/rains).

1.2 Scenario analysis and optimisation

The RAINS model framework makes it possible to estimate, for a given energy- and agricultural scenario, the costs and environmental effects of user-specified emission control policies (the "scenario analysis" mode), see Figure 1.2. Furthermore, a non-linear optimisation mode can be used to identify the cost-minimal combination of emission controls meeting user-supplied air quality targets, taking into account regional differences in emission control costs and atmospheric dispersion characteristics. The optimisation capability of RAINS enables the development of multi-pollutant, multi-effect pollution control strategies. In particular, the optimisation can be used to search for cost-minimal balances of controls of the six pollutants (SO_2, NO_x, VOC, NH_3, primary $PM_{2,5}$, primary $PM_{10-2.5}$ (= PM coarse)) over the various economic sectors in all European countries that simultaneously achieve user-specified targets for human health impacts (e.g., expressed in terms of reduced life expectancy), ecosystems protection (e.g., expressed in terms of excess acid and nitrogen deposition), and maximum allowed violations of WHO guideline values for ground-level ozone, etc. (Figure 1.2).

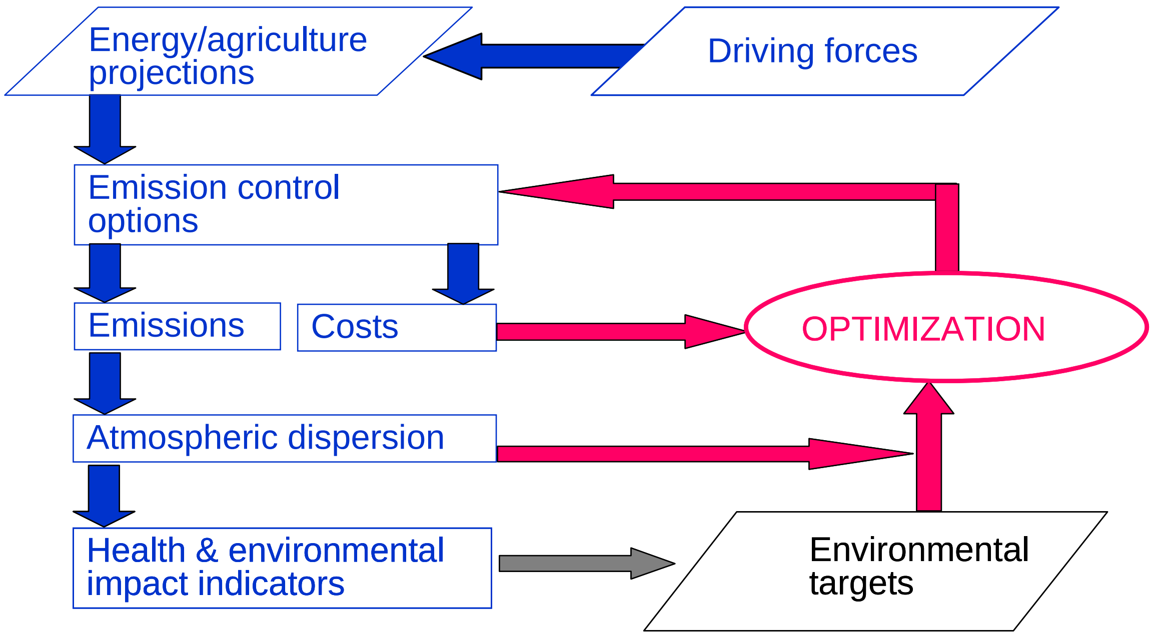

Figure 1.2: The iterative concept of the RAINS optimisation.

The RAINS model started to interest the negotiators acting within the Convention on Long-range Transboundary Air Pollution after its capabilities were extended from the initial "scenario analysis" to an "optimisation" mode. While the scenario analysis mode could be used to illustrate the economic and environmental consequences of an exogenously assumed pattern of emission controls, the optimisation feature allowed the systematic identification of the least-cost allocation of emission controls that meet exogenously determined environmental targets.

With the scenario mode, the number of "what-if" scenarios that could be explored with the RAINS model was limited, which made it impossible to fully explore the consequences of even the most important permutations of emission control measures in all economic sectors of the (up to) 48 Parties. Thus, in practice the scenarios addressed a limited number of technology-related emission control rationales, but could not add to a systematic analysis of environmentally driven emission control strategies that were in the focus of the Convention after the NO_x protocol. Although the main feature of the scenario mode was the assessment of the environmental effects of emission controls, their quantification was hampered by methodological problems in the spatial downscaling of the impact assessment, which did not allow predicting effects for specific ecosystems. Consequently, the pure scenario analysis provided only limited insight to negotiators who had to find distributions of emission control obligations across countries that were acceptable to all Parties.

The situation changed as soon as the optimisation feature of the RAINS model was developed, which made it possible to identify distributions of abatement burdens across Parties that were most "efficient" according to a selected rationale. The RAINS optimisation provided the ideal complement to the "critical loads" concept that has been accepted by the Convention since the First NO_x Protocol as a rationale for future emission control agreements, because it allowed determining the least-cost allocation of measures that would achieve environmental targets established in terms of critical loads. Thus, the optimisation concept became an important element of a "science based" rationale that was desired as a basis for the coming emission reduction accords. By calculating country- and sector-specific reduction requirements for any exogenously specified environmental target, the RAINS optimisation provided results that were of immediate relevance to the negotiators because they met the spatial and temporal scales that were relevant for decision makers. The optimisation was also attractive because, while striving for a common target (equal environmental improvement for all Parties), it considered the environmental and economic differences between Parties that lead to objectively justifiable differences in abatement efforts. Resulting inequities in abatement burdens were based on scientifically determined differences in environmental sensitivities, atmospheric dispersion characteristics or emission source structures. Thus, the negotiators could focus their negotiations on the ambition level of their environmental objectives, the political acceptability of the implied costs and their distribution across Parties, while they could leave technicalities (the quantification of objective differences between countries) to the formal model. The model was seen as a common knowledge base, which allowed negotiators to focus on the policy issues ("Let's put the facts on the table, we will fight about politics later.")

It is also important that the optimisation problem as set up in the RAINS model does not provide an absolute and unique answer to the air pollution problem. The actual results of an optimisation run depend on the environmental objectives (e.g., the acceptable environmental risk) as established by the negotiators, the goal function (minimization of total emission control costs), and the problem framing (e.g., the exclusion of changes in the energy systems, which cannot be directly influenced by environmental policies in Europe). All these settings are subject to negotiations, and the optimisation results are critically influenced by the policy choices on these issues. Thus, the RAINS model does not internalise policy choices, but deliberately leaves room for decisions of negotiators.

It is envisaged that now, with the inclusion of fine particulate matter and the complex interactions of the primary and secondary precursor emissions, a systematic search for effective solutions will be even more attractive.

1.3 System boundaries

It is at the heart of integrated assessment models to achieve integration by including as many aspects of pollution control as possible in order to gain comprehensive insights into the full range of issues related to the strategies under consideration. However, it is also crucial to keep integrated assessment models manageable in order to facilitate the direct interaction with decision makers in the analysis of a large number of alternatives in a timely manner. Thus, it is the art of integrated assessment modelling to strike the right balance between a larger range of integration on the one side and practical manageability (for modellers) and transparency (for users) on the other.

Over time, the RAINS model has included a large number of aspects of air pollution, and is now a powerful tool for providing policy relevant insight into many facets of air pollution control. However, deliberate decisions were taken by the developers of RAINS to keep certain aspects outside the model, partly because they are of less relevance than other aspects, and partly because an appropriate treatment of these issues would dramatically increase the complexity of the overall RAINS model and thus seriously compromise its performance and transparency. Nevertheless, it is recognized that many aspects that are presently not hard-wired into RAINS are important.

This applies particularly to the assessment of ancillary benefits, to the monetary evaluation of benefits and to emission control options that imply substantial structural changes in the economy (or deviations from the baseline assumptions about economic development). With the tightening stringency of emission control strategies over time, it becomes increasingly important to treat these issues properly in order to obtain a full picture of costs and benefits of possible policy action.

1.3.1 Climate change, energy and transport

In response to these needs, a number of studies were made to develop 'soft links' between RAINS and other models that treat these issues. For instance, numerous analyses were carried out that explore the influence of alternative implementation options of the Kyoto Protocol on air pollution control strategies. This was achieved through a linkage between alternative projections of energy development from the PRIMES model with RAINS. These studies concluded that, compared to a "pre-Kyoto" baseline projection, Kyoto-compliant energy structures would reduce costs for meeting the National Emissions Ceilings Directive by up to 40 percent (Syri *et al.*, 2001). It is perfectly technically feasible – and definitely instructive - to conduct similar assessments for more recent views on Kyoto and post-Kyoto implementation options, and for the environmental targets discussed under CAFE for periods beyond 2010.

At the same time, legislation on air pollution control (e.g., the National Emission Ceilings Directive and the Large Combustion Plant Directive) might directly or indirectly influence the costs for certain modes of energy production and conversion, which could in turn have some bearing on development of the energy system. An analysis along these lines was conducted with the PRIMES model, using input from the RAINS model, during the development of the preliminary baseline scenario for the DG-TREN Energy Outlook 2030. Again, it would be instructive to repeat such an analysis for the environmental targets discussed under CAFE for the time beyond 2010, and it is perfectly technically feasible to do so.

Of special importance in this respect is the future development of the transport sector. At the moment transport projections are included on a more aggregated level in the PRIMES model, and all the soft-link possibilities between RAINS and PRIMES that are discussed above are also applicable to transport scenarios. In addition, the TREMOVE transport model is now further developed, and it is expected that the revised version of TREMOVE will provide a range of detailed transport scenarios with different implications for emissions of greenhouse gases and other air pollutants. Data transfer between the old version of TREMOVE and RAINS was possible, and it is expected that the updated TREMOVE model could only improve in this respect.

These technical interface possibilities between RAINS, PRIMES and TREMOVE will allow a comprehensive and quantitative assessment of the interactions between air pollution and climate change policies in the energy sector. It needs to be decided with the CAFE secretariat at which point of the scenario analysis within CAFE such analyses would be most instructive.

1.3.2 Air pollutants and greenhouse gases

There is a growing and multi-faceted body of scientific evidence that many conventional air pollutants also act as greenhouse gases. As pointed out in the Third Assessment Report (TAR) of the Intergovernmental Panel on Climate Change (IPCC, 2001), some of the conventional air pollutants such as tropospheric ozone, SO_2, carbonaceous particles (black carbon and organic carbon) have a direct influence on radiative forcing, but are not accounted for in the Kyoto Protocol (Figure 1.3). For instance, TAR estimates a positive radiative forcing of +0.35 (±43%) W/m^2 for the changes in tropospheric ozone between 1850 and the early 1990s, compared to +0.48 W/m^2 for methane (CH_4)

and +1.46 W/m^2 for carbon dioxide (CO_2). The median of direct forcing of black carbon from fossil fuels is estimated at +0.20 W/m^2. At the same time, other air pollutants exert negative forcing, e.g., the direct effect of sulphate aerosols is estimated at –0.40 W/m^2, of fossil fuel organic carbon –0.10 W/m^2 and biomass burning aerosols –0.20 W/m^2. In developing air pollutant emission control strategies within CAFE it will be important to consider the net effects of proposed policies. The RAINS model is presently being extended to address these issues and to quantify, as far as it is possible on solid scientific grounds, the radiative effects of emission control strategies.

In addition, the IPCC also identified inter-related effects of air pollutants. For instance, precursor gases such as NO_x, VOC, CO, SO_2 and NH_3, although not radiatively active on their own, influence the radiation balance by forming radiatively active ozone and secondary aerosols. Following the progress of science, scientific work at IIASA is developing methodologies to quantify the effect of these air pollutants in the radiation balance. If accepted by scientific peers, the outcome of this activity will be linked to the RAINS model so that it can be used for scenario analysis for the work under CAFE.

It should be mentioned that there are also other, possibly important, linkages between air pollution and climate change. For instance, emissions of NO_x do not only have the indirect (positive) radiative forcing via their contribution to the formation of ozone and secondary aerosols, they also chemically influence the abundance of OH radicals in the atmosphere and thus indirectly affect the lifetime of methane, which acts as a potent greenhouse gas. Via this pathway, NO_x exerts a negative radiative forcing. At the same time, emissions of NO_x lead to increased deposition of nitrogen compounds on the earth surface, which in turn act as fertilizer to plant growth leading to higher uptake of CO_2 by plants from the atmosphere (equivalent to a negative radiative forcing). While it might be difficult to quantify exactly the net effect of these mechanisms on radiative forcing based on solid scientific understanding within the project time, it is proposed that these effects will be qualitatively discussed in the reports prepared for the CAFE programme.

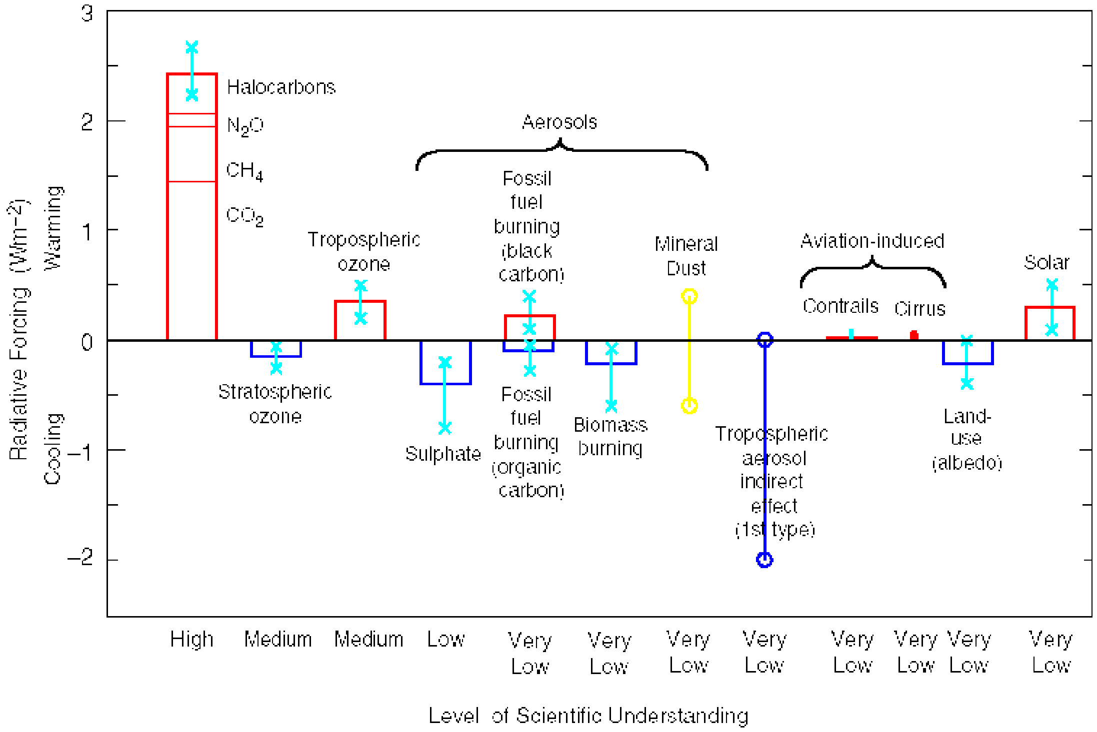

Figure 1.3: Global annual mean radiative forcing due to various agents for the period pre-industrial to present. Source: IPPC, 2001

Another potentially relevant issue concerns the influence of hemispheric emissions of CH_4 and CO on ozone background levels. It is clear that, in addition to European NO_x and VOC emissions, CH_4 and CO are also important precursors of tropospheric ozone, and that changes in hemispheric emissions of these substances will influence background ozone levels in Europe. While, based on recent analysis at IIASA, the increases in global CH_4 and CO emissions projected by the IPCC-SRES scenarios appear unrealistically high, there exists a significant potential for controlling these greenhouse gases through a variety of measures, often at low costs (AEAT, 1998). It remains to be quantified to what extent such emission controls, if taken in a coordinated fashion at the hemispheric scale, could substitute for (expensive) further reductions of NO_x and VOC emissions in Europe in order to bring ozone levels in Europe towards the EU long-term targets. IIASA and MET.NO are currently exploring this issue in more detail with a view to its potential inclusion in the RAINS analysis.

1.3.3 Agricultural policies

Earlier analyses with the RAINS model indicated that emissions from agricultural activities make important contributions to a range of air quality related problems. Despite the limits imposed by the Emission Ceilings Directive, emissions of ammonia will become the dominating source of nitrogen deposition in many areas in Europe. It will be difficult to approach the ultimate target of fully bringing deposition of acidifying and eutrophying substances below the critical loads without further reductions in these emissions. Not yet widely recognized, emissions of ammonia are an important precursor to the formation of secondary inorganic aerosols (ammonium sulphates and ammonium nitrates), which constitute a major fraction of PM2.5 in Europe (Daemmgen, 2002). Thus, in addition to the technical control measures on ammonia emissions that are considered in the RAINS model, analysis in CAFE should address the potential contributions of changes in agricultural policies on the cost-effective achievement of air quality targets in Europe.

While the RAINS model cannot develop scenarios of agricultural activities, such scenarios, in particular those developed in the context of the reform of the Common Agricultural Policy, if supplied by other sources such as DG-AGRI, can be introduced into the RAINS model and then used to assess their potential implications for clean air policies in Europe. The interface between the CAPRI and RAINS models will facilitate such analysis.

In theory, there are also feedbacks from low air quality on agricultural productivity. Most prominently, high ozone levels might cause damage to agricultural crops in Mediterranean countries, depending on irrigation conditions. In principle, high rates of nitrogen deposition have a fertilization effect on agricultural crops, but the magnitude of such impacts has not yet been quantified. Increased acid deposition on agricultural soils is usually compensated by the present fertilization practices, so that this linkage seems of less relevance. With the RAINS model providing estimates of ozone fields in the rural areas, an important piece of information will be available to assess impacts of ozone on agricultural crops. A precise quantification of the damage will depend on the availability of appropriate dose-response curves and their acceptance by the scientific community.

Also in the agricultural field potentially important linkages between air pollution and climate should not be ignored. It has been shown by IIASA researchers that critical interactions between the control of ammonia and greenhouse gases such as CH_4 and nitrous oxide (N_2O) exist (Brink *et al.*, 2001). For instance, application of all technical measures to reduce NH_3 emissions in Europe (achieving a 36 percent reduction in NH_3) would lead to 15 percent higher N_2O emissions, but at the same time reduce CH_4 emissions by about two percent. Work is underway to include these calculations into the routine RAINS scenario analysis.

1.3.4 Water and soil quality

Measures to reduce emissions of air pollutants have a range of impacts on the quality of waters and soils as well as for a range of other environmental endpoints. It is important that in the process of policy deliberations such effects are not forgotten and that they are clearly quantified as far as technically and scientifically possible and justifiable based on their magnitudes.

The RAINS model includes all the routines necessary to quantify improvements in water and soil quality due to the reduction of acidifying and eutrophying deposition to the extent that is possible on solid scientific ground. In particular, in agreement with the scientific community studying these ecological effects, the RAINS model incorporates the ‘critical loads’ concept. Thus on a routine basis it allows, for any emission control scenario under consideration, the quantification of deposition in excess of critical loads and the area/share of ecosystems that are protected from acidification/eutrophication according to present scientific knowledge. IIASA is closely following progress in the scientific understanding of dynamic acidification processes. If common understanding develops, its implications can be evaluated with the information provided by RAINS.

It is understood that reduced emissions of air pollution will also lead to a range of other environmental improvements that are presently not fully considered in RAINS. The study on European environmental priorities conducted for the European Commission (RIVM *et al.*, 2001) attempted to quantify such effects to the maximum possible extent. In particular, impacts were found for biodiversity and the eutrophication of seas (notably of the Baltic). For biodiversity, the scientific peer review undertaken for the study concluded, however, that an accurate quantification of the effects remains problematic without further original research. The report also pointed out that for the eutrophication of seas through nitrates and phosphates, nitrogen input from the atmosphere is only one pathway, and that input from rivers dominate the total budget. It was also found that measures for controlling pollution discharges to water do not have major side effects on air pollution.

It is therefore proposed for the purposes of the assessment in CAFE

- for biodiversity to rely on the assessment conducted in the context of the "Priority Study" and, if necessary, interpolate or extrapolate quantitative findings of this study with data of selected CAFE scenarios, and
- for the eutrophication of seas to quantify the atmospheric deposition of nitrogen to the regional seas for selected scenarios. Keeping in mind that other sources might make the dominating contributions, no full assessment of eutrophication of seas is suggested.

1.4 The role of cost-benefit and multi-criteria analysis

1.4.1 Monetary evaluation of benefits

It is recognized that a monetary evaluation of the benefits of emission control strategies could provide essential information that helps decision makers in striking the right balance between environmental ambition and the economic implications.

However, in practice monetary evaluations of environmental benefits are loaded with a wide range of problems that make their results in many cases rather controversial if they are used in a policy context. One type of difficulty is related to practical problems with actually quantifying environmental damage from air pollution on a solid scientific basis. Often the scientific communities working in such fields do not feel that their present insights are good enough to allow credible quantifications (e.g., for ozone damage to plants, to quantify the effects of acidification on vegetation, etc.). A second complication is caused by the difficulties with attributing economic values to certain non-market goods, most notably to the values of human life and ecosystems. Although a variety of economic approaches exist that indirectly distil such values from observations, experience shows that the results of such analyses often remain controversial, and that a heavy reliance of strategy development on such estimates does not facilitate ultimate consensus between parties with conflicting interests.

While recognizing the potential usefulness of such economic evaluation techniques, the developers of the RAINS model have decided not to internalise such controversial techniques into the model, but restrict the formal model calculations to fields where general consensus (about physical processes, economic evaluations of costs, etc.) exists. Still, the modellers consider it useful to interface (soft-link) RAINS with other tools that address the economic and monetary evaluation of benefits and via this pathway provide such information to users who want to see it. Thus, RAINS provides important scenario-specific information to frameworks that estimate monetary benefits. Such information includes, e.g., fields of ambient levels and deposition of various air pollutants over all of Europe with a 50*50 km resolution, levels of pollution in urban areas, the size and age structure of population exposed to different pollution levels in Europe, the extent (and possibly types) of ecosystems that are exposed to various pollution levels, etc.

In the past for the analysis conducted for the Emission Ceilings Directive, such an interface was successfully operated with AEA-Technology (IIASA and AEAT, 1999) for an analysis of the monetary benefits of the various emission control strategies, following the methodology developed under the EXTERNE project. For the present proposal, IIASA will (re-) connect to such assessment tools and provide them with the required information for carrying out their tasks.

1.4.2 Cost-benefit analysis

The RAINS model uses optimisation techniques to identify emission control strategies that are efficient according to selected criteria. Traditionally, a cost-effectiveness approach was used that determined the least-cost combinations of emission control measures that achieve user-defined environmental air quality targets. In the iterative processes of recent RAINS policy applications, decision makers specified a series of environmental constraints with different ambition levels, and the optimisation routine of RAINS was used to identify the internationally cost-optimal solutions to meet these targets.

The cost minimization concept presently implemented in RAINS is only one of the conceivable optimisation criteria. Early experiments with RAINS explored the practical usefulness of alternative optimisations that, e.g., minimized environmental impacts for a total budget constraint. Other integrated assessment models, e.g., the Imperial College's ASAM model (Warren and ApSimon, 2000) tested further concepts. Consultations with decision makers, however, led to the conclusion that the cost-effectiveness principle, materialized through the cost minimizing optimisation as implemented in RAINS, met best the needs of the actual setting of international environmental policy in Europe.

This does not mean that alternative optimisation concepts could not be useful. In particular, a fully internalised cost-benefit approach is suggested from time to time by various stakeholders as the theoretically most appropriate concept. In such a case, the environmental ambition level would not be set "externally" by decision makers, but determined by the model through balancing the costs of measures against the benefits of actions expressed in monetary terms. Given the existing disagreement about the monetary quantification of benefits and the difficulties in quantifying certain benefits at all, the developers of the RAINS model have decided, for the time being, not to embark on a fully internalised cost-benefit optimisation analysis. Instead, the RAINS developers explicitly foresee a role for decision makers within the iterative cycle of model applications in a practical policy context, where decision makers themselves decide about the acceptable balance between environmental ambition and incurred costs. This issue was discussed with all stakeholders at length at an earlier session of the CAFE Steering Group, and sufficient time is now reserved in the CAFE work plan (and in the call for tenders) for conducting such iterative interaction between decision makers and modellers.

It should be mentioned that the decision not to include a full cost-benefit analysis in RAINS was not taken for technical reasons (the RAINS framework could be easily extended to allow such analysis), but primarily for the conceptual arguments presented above. If, in the course of the deliberation of the CAFE programme, consensus would emerge about the usefulness of a full cost-benefit analysis, the RAINS model could be adapted accordingly.

1.4.3 Multi-criteria analysis

As an alternative concept, the developers of the RAINS model opted for a multi-criteria analysis. There are many different concepts of such multi-criteria assessments proposed in the literature, ranging from simple presentation schemes of multiple model output to full-fledged multi-criteria optimisation analyses using sophisticated techniques to implicitly derive preference structures of the decision makers that are often hidden. Scientific activities at IIASA have a long-standing record in playing a leading role in multi-criteria analyses, so that the choice of the appropriate method in the context of RAINS was not limited by technical constraints (expertise on and availability of methods), but determined by considerations of usefulness in the practical policy context of RAINS.

At the moment, the RAINS model provides a large range of different model results that provide useful information to decision makers who have to decide about preferred emission control strategies. Among these results, the model delivers for a given emission control scenario

sectoral emission reductions for the various pollutants in the various countries,

sectoral emission reduction costs for the individual pollutants by country,

listings of technological means that need to be adopted in the various countries and economic sectors in order to meet the environmental targets,

emissions and emission control costs aggregated to countries, in absolute terms and in relation to a base year,

fields of ambient concentrations of ozone, PM10 and PM2.5 across Europe with a 50*50 km resolution,

estimates of ozone, PM10 and PM2.5 concentrations in urban areas (pending the results of the CITY DELTA project),

fields of acid deposition, distinguishing sulphur, oxidized and reduced nitrogen compounds in 50*50 km^2,

fields of nitrogen deposition, distinguishing oxidized and reduced nitrogen compounds in 50*50 km^2,

accumulated excess deposition of acidifying compounds exceeding the critical loads of all ecosystems in a grid cell in 50*50 km^2,

the area/percentage of ecosystems with acid/nitrogen deposition above their critical loads, with a 50*50 km^2 resolution,

excess deposition for selected ecosystems,

number of people that are exposed in rural/urban areas to PM/ozone concentrations above selected threshold values (e.g., WHO guideline values),

- loss in statistical life expectancy due to PM pollution, per country/grid cell, etc.

Additional and more detailed output can be produced on demand.

It is foreseen that the reports produced for the CAFE baseline and policy scenarios will provide all this information to decision makers and the CAFE Steering Group. Experience of earlier policy applications suggests, however, that in many cases decision makers tend to be overwhelmed by the wealth of information, and that thoroughly aggregated indicators can be more efficient in allowing practical comparisons of alternative policy scenarios. The RAINS model provides all technical capabilities to aggregate results in any desired way and/or to produce graphical representations (diagrams) of selected key results. In practice, however, it turned out that the optimal set of reported results only emerges through close interaction with the users of the output, which in the context of CAFE would be the Commission, the representatives of Members States and other stakeholders participating in the CAFE Steering Group. IIASA is ready to develop appropriate and efficient forms of model output in a multi-criteria setting together with the model users and thus does not want to provide, without consulting with the users, definite lists of output formats at this stage.

2 Modelling of driving forces

2.1 Anthropogenic driving forces for air pollution emissions

Anthropogenic activities such as energy consumption, industrial activities and agriculture are major driving forces of emissions of air pollutants. Their future development has a strong influence on the level of future emissions and on the potential and costs for maintaining emissions at environmentally acceptable levels. Unfortunately, it is an ambitious task to accurately model the future development of anthropogenic economic activities at the level of detail that is required for the assessment of air pollution. A number of economic theories compete in this field, and their modelling entails complex approaches and a variety of detailed assumptions, which are difficult to quantify on an undisputed basis. Thus, as a first choice, it has been decided not to embark with the RAINS model on the modelling of future economic activities, but to derive projections from other sources as an exogenous input to the RAINS model.

However, numerous scenario studies with the RAINS model have shown that modifications in these exogenous drivers (e.g., energy consumption, agricultural activities) yield in many cases larger and more cost-effective potentials for reducing emissions than the application of add-on/end-of-pipe emission control technologies (Syri *et al.*, 2001; Barkman *et al.*, 2003; Rentz *et al.*, 1994). For these studies, interfaces between the RAINS model and specialized energy models (PRIMES, TIMER, EFOM, MESSAGE) have been developed that allow the import of alternative energy scenarios into the RAINS database.

The strong impact of alternative economic projections on air pollution raises two important issues for the RAINS calculations: First, policy interventions that influence such driving forces could turn out to be a very cost-effective means for controlling air pollution, and an integrated assessment needs to take this potential into account. Second, when developing baseline projections of future air quality and searching for cost-effective emission control strategies, uncertainties in these projections cannot be ignored and strategies need to be found that are robust against these uncertainties in the input drivers.

To address the first concern, a rule-based software interface between the PRIMES energy model and the RAINS model has been developed, which requires only minimal additional expert knowledge. This interface opens the possibility for the analysis of a larger number of energy scenario variants. Similar action is underway to convert alternative projections of agricultural activities developed with the CAPRI model of the University of Bonn into the RAINS databases. Comparative air quality analyses for alternative economic projections will identify factors and structural measures in the economy that have beneficial impacts on air pollution control strategies.

It remains difficult to interpret any of the projections as an accurate prediction of future development. Thus, any calculation of an emission (control) scenario based on a particular energy or agricultural projection is loaded with significant uncertainties. In many cases, the uncertainties resulting from the underlying exogenous assumptions (e.g., on energy prices, economic development, carbon prices, etc.) dominate uncertainties associated with other parts in the chain of RAINS model calculation (Suutari *et al.*, 2001).

2.2 Projections of emission generating activities

Since it is hard to predict some of the important determinants of future emissions on a reliable basis, the RAINS analysis will focus on the robustness of model results in view of these unavoidable uncertainties. For this purpose, the RAINS databases for the CAFE policy analysis include multiple baseline projections on energy use and agricultural activities:

- A Europe-wide consistent view of energy development with certain assumptions on climate policies (as produced by the PRIMES energy model).
- As a variant, a Europe-wide consistent view of energy development without climate policies. For this purpose, RAINS uses the Energy 2030 outlook of DG-TREN.
- A compilation of official national projections of energy development with climate policies that reflect the perspectives of the individual governments of Member States. By their nature, there will be no guarantee of international consistency in the main assumptions across countries (e.g., economic development, energy prices, use of flexible mechanisms for the Kyoto Protocol, assumptions on post-Kyoto regimes, etc.).

For agriculture, RAINS will use

- a set of Europe-wide consistent projections of agricultural activities without CAP reform, and
- a compilation of national projections of activities supplied by Member States.
- In addition, it is foreseen that a 'CAP reform' projection will be made available by DG-AGRI once the policy plans are agreed upon.

The policy analysis will then focus on environmental targets that lead to further improvements of air quality and will explore the implications of alternative baseline projections on achieving these targets. Thus, there is no need to reach full consensus of all stakeholders on all assumptions of each baseline projection, as long as overall plausibility and consistency is maintained.

To the extent available, alternative projections of drivers have been implemented in the on-line version of the RAINS model (http://www.iiasa.ac.at/web-apps/tap/RainsWeb/RainsLogin.htm) and are ready for analysis.

These baseline projections include assumptions about the general economic development, such as

GDP growth rates for the different economic sectors,

energy (specifying demand and supply of different fuel types in the various economic sectors),

agricultural production (e.g., number of animals),

transport (e.g., fuel consumption by vehicle types, off-road activities, etc.) and

- industrial production (distinguishing different kinds of goods and their production methods).

The baseline projections will be based on full compliance with existing and adopted national and Community legislation (e.g., the Air Quality, LCP and NEC directives). Thus, the projections must comply with the targets that the EU Member States have ratified in the Kyoto Protocol. However, in order to understand the significance of the Kyoto Protocol, a scenario will be prepared where the Kyoto constraint is not binding. This is because it is not known at the moment to what extent the Member States will take advantage of the flexible mechanisms (International Emissions Trading, Joint

Implementation and Clean Development Mechanisms) of the Kyoto Protocol and what the consequent effects on the fuel mix (and thus air pollution) are likely to be. Some other alternative scenarios are also conceivable for the CAFE baseline analysis.

As it is possible that the Member States and Accession Candidate Countries have slightly different views on the driving forces of emissions, it is important to include such views when the CAFE baseline is developed. However, it needs to be emphasised that such alternative views need to be consistent with the national, community-wide and international obligations that the Member State has undertaken. In other words, the possible alternative baseline that is suggested by a Member State or Accession Candidate Country needs to be compliant with, e.g., NEC, LCP and Air Quality directives, as well as the Kyoto Protocol.

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3 Modelling of emissions

3.1 The objectives of emission and control cost calculations within the framework of an integrated assessment model

One of the central objectives of integrated assessment models is to assist in the cost-effective allocation of emission reduction measures across various pollutants, several countries and different economic sectors. Obviously, this task requires consistent information about the costs of emission control at the individual sources, and it is the central objective of this cost module to provide such information.

The optimal allocation of emission control measures between countries is crucially influenced by differences in emission control costs for the individual emission sources. It is therefore of utmost importance to identify systematically the factors leading to differences in emission control costs among countries, economic sectors and pollutants. Such differences are usually caused, *inter alia,* by variations in the composition of the various emission sources, the state of technological development and the extent to which emission control measures are already applied.

3.2 Aggregation of emission sources

Emissions of air pollutants are released from a large variety of sources with significant technical and economic differences. Conventional emission inventory systems, such as the CORINAIR inventory of the European Environmental Agency, distinguish more than 300 different processes causing various types of emissions.

In the ideal case, the assessment of emissions and the potential and costs for reducing emissions should be carried out at the very detailed process level. In reality, however, the necessity to assess abatement costs for all countries in Europe, as well as focus on emission levels in 10 to 20 years from now, restricts the level of detail which can be maintained. While technical details can be best reflected for individual (reference) processes, the accuracy of estimates on an aggregated national level for future years will be seriously hampered by a general lack of reliable projections of many of these process-related parameters (such as future activity rates, autonomous technological progress, etc.). For an integrated assessment model focusing on the pan-European scale it is therefore imperative to aim at a reasonable balance between the level of technical detail and the availability of meaningful data describing future development, and to restrict the system to a manageable number of source categories and abatement options.

3.3 Criteria for the aggregation

For the RAINS model, an attempt was made to aggregate the emission producing processes into a reasonable number of groups with similar technical and economic properties. Considering the intended purposes of integrated assessment, the major criteria for aggregation were:

- The importance of the emission source. It was decided to target source categories with a contribution of at least 0.5 percent to the total anthropogenic emissions in a particular country.
- The possibility of defining uniform activity rates and emission factors.

- The possibility of constructing plausible forecasts of future activity levels. Since the emphasis of the cost estimates in the RAINS model is on future years, it is crucial that reasonable projections of the activity rates can be constructed or derived.
- The availability and applicability of "similar" control technologies.
- The availability of relevant data. Successful implementation of the module will only be possible if the required data are available.

It is important to define carefully the appropriate activity units. They must be detailed enough to provide meaningful surrogate indicators for the actual operation of a variety of different technical processes, and aggregated enough to allow a meaningful projection of their future development with a reasonable set of general assumptions.

The RAINS source structure distinguishes emission categories for several stationary and mobile combustion sources, which are split by relevant activities, and also a number of other non-combustion sectors. Some categories are further disaggregated to distinguish, for example, between existing and new installations in power plants, or between tyre and brake wear for non-exhaust emissions from transport (for a full list of RAINS sectors see Annex 1).

The sectoral structure of the RAINS model is not directly compatible with that of CORINAIR or the UNECE reporting standard (NFR – **N**omenclature **F**or **R**eporting) (UNECE, 2002). In several cases, the relation between RAINS sectors and the other sectoral classification schemes can be established only for a primary sector, i.e., the sum of all RAINS categories for power and district heating plants can only be compared with the sum of several SNAP entries. RAINS contains a feature to aggregate/display emissions into the CORINAIR SNAP level 1 as well as NFR level 1 and 2.

3.4 Emission factors

RAINS estimates emissions based on activity data, uncontrolled emission factors, the removal efficiency of emission control measures and the extent to which such measures are applied:

$$E_i = \sum_{j,k,m} E_{i,j,k,m} = \sum_{j,k,m} A_{i,j,k} ef_{i,j,k} (1 - eff_m) X_{i,j,k,m} \quad (1)$$

where:

i,j,k,m	Country, sector, activity type, abatement technology;
E_i	Emissions in country *i*;
A	Activity (level) in a given sector, e.g. coal consumption in power plants;
ef	"Raw gas" emission factor;
eff_m	Reduction efficiency of the abatement option *m*, and;
X	Actual implementation rate of the considered abatement, e.g., fraction of total coal used in power plants that are equipped with electrostatic precipitators.

With this approach, emission factors are the key to assess emissions accurately. For RAINS it has been decided to identify, as far as possible, the main factors that could lead, for a given source category, to justified differences in emission factors across countries. The aim has been to collect country-specific information to quantify such justifiable deviations from values reported in the general literature. When this was not possible or when a source category makes only a minor contribution to total emissions, emission factors from the literature were used. The approach for establishing country-specific emission factors depends on the pollutant under consideration, and details are provided in the pollutant-specific documentation (Cofala and Syri, 1998a; Cofala and Syri, 1998b; Klimont *et al.*, 2000; Klimont *et al.*, 2002).

For the earlier analysis for the Emission Ceilings Directive, it was possible, in most cases, to limit discrepancies between RAINS emission estimates and national inventories to a few percent. Only a handful of cases remained where larger discrepancies could not be resolved in discussions with national experts. Where national estimates could not be reproduced with a plausible set of data according to Equation 1, RAINS used its own estimates to maintain international consistency, while explicitly stating the points of disagreement in the scenario and policy analysis reports (see, e.g., Amann *et al.*, 1999).

3.5 Emission projections

RAINS estimates future emissions according to Equation 1 by varying the activity level along the projection of anthropogenic driving forces and by adjusting the implementation rate of emission control measures (*X*). With this approach, the "uncontrolled" emission factor remains unchanged, and any reduction in emissions is attributed to the implementation of control measures (*X*), for which costs are estimated in a further step. In the optimisation mode, the implementation rates (*X*) become the decision variables of the optimisation problem.

3.6 Uncertainties

A methodology has been developed to estimate uncertainties of emission calculations based on uncertainty estimates for the individual parameters of the calculation (Suutari *et al.*, 2001). It was found that uncertainties in modelled national emissions of SO_2, NO_x, NH_3 in Europe typically lie in the range between 10 and 30 percent (Table 3.1: Expected emissions, emission uncertainties and correlation between SO2 and NOx emissions in 1990.

, Table 3.2) . In general, the uncertainties are strongly dependent on the potential for error compensation. This compensation potential is larger (and uncertainties are smaller) if calculated emissions are composed of a larger number of similar-sized source categories, where the errors in input parameters are not correlated with each other. Thus, estimates of national total emissions are generally more certain than estimates of sectoral emissions.

A sensitivity analysis with respect to the uncertainty in input parameters (Table 3.3) showed that the actual uncertainties are critically influenced by the specific situation (pollutant, year, country). Generally, however, the emission factor is an important contributor to the uncertainty in estimates of historical emissions, while uncertainty in the activity data dominates the future estimates.

Table 3.1: Expected emissions, emission uncertainties and correlation between SO_2 and NO_x emissions in 1990.

	SO_2		NO_x		SO_2/NO_x	NH_3	
Country	Expected value (kt)	95 percent confidence interval	Expected value (kt)	95 percent confidence interval	Correlation	Expected value (kt)	95 percent confidence interval

Albania	72	±10%	24	±12%	0.08	32	±23%
Atlantic Ocean	641	±19%	911	±26%	0.29	n.a.	n.a.
Austria	93	±9%	192	±10%	0.03	77	±10%
Baltic Sea	72	±19%	80	±26%	0.29	n.a.	n.a.
Belarus	843	±12%	402	±11%	0.16	219	±17%
Belgium	336	±13%	351	±13%	0.04	97	±11%
Bosnia-Herzegovina	487	±19%	80	±15%	0.17	31	±16%
Bulgaria	1842	±21%	355	±13%	0.06	141	±18%
Croatia	180	±10%	82	±14%	0.04	40	±16%
Czech Rep.	1873	±20%	546	±18%	0.11	107	±14%
Denmark	182	±10%	274	±9%	0.28	77	±12%
Estonia	275	±18%	84	±13%	0.12	29	±17%
Finland	226	±8%	276	±9%	0.06	40	±10%
France	1250	±6%	1867	±11%	0.06	810	±11%
Germany, New Länder	4438	±16%	702	±15%	0.18	201	±16%
Germany, Old Länder	842	±6%	1960	±11%	0.07	556	±11%
Greece	504	±7%	345	±8%	0.10	80	±21%
Hungary	913	±16%	219	±12%	0.06	120	±18%
Ireland	178	±7%	113	±9%	0.21	127	±13%
Italy	1679	±11%	2037	±9%	0.10	462	±14%
Latvia	121	±8%	117	±11%	0.08	43	±16%
Lithuania	213	±12%	153	±11%	0.11	80	±16%
Luxembourg	14	±14%	22	±12%	0.02	7	±15%
FYR Macedonia	107	±22%	39	±22%	0.09	17	±17%
Mediterranean Sea	12	±19%	13	±26%	0.29	n.a.	n.a.
Rep. of Moldova	197	±10%	87	±10%	0.17	47	±14%
Netherlands	201	±10%	542	±9%	0.05	233	±13%
North Sea	439	±19%	639	±26%	0.29	n.a.	n.a.
Norway	52	±17%	220	±11%	0.02	23	±14%
Poland	3001	±11%	1217	±12%	0.27	505	±17%
Portugal	343	±8%	303	±10%	0.14	77	±10%
Romania	1331	±17%	518	±11%	0.07	292	±15%
Russia Kaliningrad	44	±11%	29	±11%	0.13	11	±14%
Russia, Kola-Karelia	739	±18%	111	±12%	0.06	6	±14%
Russia, remaining area	3921	±8%	3126	±11%	0.06	1221	±15%

Russia, St. Petersburg	308	±12%	221	±11%	0.13	44	±14%
Slovakia	548	±12%	219	±12%	0.09	60	±19%
Slovenia	200	±20%	60	±15%	0.09	23	±19%
Spain	2189	±12%	1162	±9%	0.06	352	±15%
Sweden	117	±9%	338	±10%	0.05	61	±9%
Switzerland	43	±9%	163	±13%	0.08	72	±13%
Ukraine	3706	±9%	1888	±10%	0.15	729	±15%
United Kingdom	3812	±11%	2839	±10%	0.23	329	±12%
Serbia and Montenegro	585	±23%	211	±23%	0.09	90	±14%

Table 3.2: Expected emissions, emission uncertainties and correlation between SO_2 and NO_x emissions for the year 2010.

	SO_2		NO_x		SO_2/NO_x	NH_3	
	Expected value (kt)	95 percent confidence interval	Expected value (kt)	95 percent confidence interval	Correlation	Expected value (kt)	95 percent confidence interval
Albania	55	±9%	36	±22%	0.20	35	±23%
Atlantic Ocean	641	±28%	911	±33%	0.62	n.a.	n.a.
Austria	39	±15%	97	±12%	0.07	67	±15%
Baltic Sea	72	±28%	80	±33%	0.62	n.a.	n.a.
Belarus	494	±14%	316	±15%	0.20	163	±17%
Belgium	171	±24%	169	±16%	0.02	96	±17%
Bosnia-Herzegovina	415	±19%	60	±14%	0.23	23	±15%
Bulgaria	846	±22%	297	±17%	0.06	126	±20%
Croatia	70	±15%	91	±19%	0.04	37	±22%
Czech Rep.	336	±17%	312	±16%	0.29	108	±14%
Denmark	146	±20%	141	±10%	0.38	72	±15%
Estonia	107	±24%	49	±16%	0.20	29	±20%
Finland	137	±17%	149	±11%	0.13	31	±13%
France	574	±16%	860	±12%	0.03	780	±14%
Germany, New Länder	141	±15%	219	±12%	0.21	147	±15%
Germany, Old Länder	372	±12%	868	±12%	0.12	425	±14%
Greece	508	±13%	342	±10%	0.16	74	±33%
Hungary	227	±28%	159	±15%	0.06	137	±23%
Ireland	119	±15%	79	±10%	0.56	130	±18%
Italy	381	±22%	1013	±13%	0.06	432	±17%
Latvia	71	±10%	84	±14%	0.16	35	±22%
Lithuania	61	±16%	95	±17%	0.11	81	±17%
Luxembourg	8	±36%	10	±17%	0.01	9	±25%
FYR Macedonia	81	±20%	29	±18%	0.17	16	±23%
Mediterranean Sea	12	±28%	13	±33%	0.62	n.a.	n.a.
Rep. of Moldova	117	±11%	66	±13%	0.23	48	±19%
Netherlands	76	±21%	247	±12%	0.03	141	±15%
North Sea	439	±28%	639	±33%	0.62	n.a.	n.a.
Norway	32	±30%	178	±16%	0.06	21	±18%
Poland	1453	±15%	728	±11%	0.35	541	±14%

Portugal	195	±15%	259	±13%	0.18	73	±16%
Romania	594	±20%	458	±13%	0.08	304	±17%
Russia Kaliningrad	18	±16%	25	±18%	0.10	11	±19%
Russia, Kola-Karelia	473	±34%	86	±14%	0.03	4	±14%
Russia, remaining area	1717	±12%	2517	±15%	0.06	845	±14%
Russia, St. Petersburg	136	±18%	170	±14%	0.16	33	±14%
Slovakia	137	±13%	132	±16%	0.06	47	±19%
Slovenia	114	±30%	57	±19%	0.11	21	±22%
Spain	1006	±15%	849	±11%	0.10	383	±18%
Sweden	65	±17%	189	±12%	0.10	61	±12%
Switzerland	26	±13%	79	±13%	0.19	66	±20%
Ukraine	1506	±13%	1433	±13%	0.14	649	±14%
United Kingdom	962	±15%	1198	±11%	0.22	297	±17%
Serbia and Montenegro	269	±25%	152	±18%	0.16	82	±14%

Table 3.3: Results from a sensitivity analysis: 95 percent confidence intervals for estimates of national SO_2 and NO_x emissions in the UK.

	SO_2		NO_x	
	1990	2010	1990	2010
Activity data	±8 %	±14 %	±5 %	±8 %
Emission factors	±7 %	±6 %	±9 %	±7 %
Removal efficiency	±0 %	±3 %	±0 %	±3 %
All factors considered	±11 %	±15 %	±10 %	±11 %

3.7 References

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3.8 Annex 1: List of RAINS sectors and activities

Table 3.4: Sectors distinguished for the RAINS emission calculation

SEC_ABB	NAME	INPUT_UNIT
AGR_ARABLE	Agriculture: Ploughing, tilling, harvesting	M ha
AGR_BEEF	Agriculture: Livestock - other cattle	M animals
AGR_BURN	Stubble burning and other agr. waste	kt VOC
AGR_COWS	Agriculture: Livestock - dairy cattle	M animals
AGR_OTANI	Agriculture: Livestock - other animals (sheep, horses)	M animals
AGR_OTHER	Agriculture: Other (activity as emissions in kt)	kt
AGR_PIG	Agriculture: Livestock - pigs	M animals
AGR_POULT	Agriculture: Livestock - poultry	M animals
ARCH_P	Architectural use of paints	kt paint
AUTO_P	Manufacture of automobiles	kveh
AUTO_P_NEW	Manufacture of automobiles (new installations)	kveh
CAR_EVAP	Evaporative emissions from cars	PJ
CONSTRUCT	Construction activities	M m2
CON_COMB	Fuel production & conversion: Combustion	PJ
CON_COMB1	Fuel production & conversion: Combustion, grate firing	PJ
CON_COMB2	Fuel production & conversion: Combustion, fluidized bed	PJ
CON_COMB3	Fuel production & conversion: Combustion, pulverized	PJ
CON_LOSS	Losses during transmission & distribution of final product	PJ
DEGR	Degreasing	kt SLV
DEGR_NEW	Degreasing (new installations)	kt SLV
DOM	Combustion in residential-commercial sector (liquid fuels)	PJ
DOM_FPLACE	Residential-Commercial: Fireplaces	PJ
DOM_MB_A	Residential-Commercial: Medium boilers (<50MW) - automatic	PJ
DOM_MB_M	Residential-Commercial: Medium boilers (<1MW) - manual	PJ
DOM_OS	Domestic use of solvents (other than paint)	mln POP
DOM_P	Domestic use of paints	kt paint
DOM_SHB_A	Residential-Commercial: Single house boilers (<50 kW) - automatic	PJ
DOM_SHB_M	Residential-Commercial: Single house boilers (<50 kW) - manual	PJ
DOM_STOVE	Residential-Commercial: Stoves	PJ
DRY	Dry cleaning	kt TEX
DRY_NEW	Dry cleaning (new installations)	kt TEX
D_GASST	Gasoline distribution - service stations	PJ
D_REFDEP	Gasoline distribution - transport and depots	PJ
EXD_GAS	Extraction, proc. and distribution of gaseous fuels	kt VOC
EXD_GAS_NEW	Distribution of gaseous fuels - new mains	kt VOC

SEC_ABB	NAME	INPUT_UNIT
EXD_LQ	Extraction, proc. and distribution of liquid fuels	kt VOC
EXD_LQ_NEW	Extraction, proc., distribution of liquid fuels (incl. new (Un)Load	kt VOC
FCON_OTHN	Fertilizer use - other N fertilizers	kt N
FCON_UREA	Fertilizer use - urea	kt N
FERPRO_ALL	Fertilizer production - total	
FERTPRO	Fertilizer production	kt N
FOOD	Food and drink industry	mln POP
GLUE	Application of glues and adhesives in industry	kt VOC
IND_OS	Other industrial use of solvents	kt VOC
IND_OTH	Other industrial sources	kt VOC
IND_P	Other industrial use of paints	kt paint
INORG	Inorganic chemical industry, fertilizers and other	kt VOC
IN_BO	Industry: Combustion in boilers	PJ
IN_BO1	Industry: Combustion in boilers, grate firing	PJ
IN_BO2	Industry: Combustion in boilers, fluidized bed	PJ
IN_BO3	Industry: Combustion in boilers, pulverized	PJ
IN_OC	Industry: Other combustion	PJ
IN_OC1	Industry: Other combustion, grate firing	PJ
IN_OC2	Industry: Other combustion, fluidized bed	PJ
IN_OC3	Industry: Other combustion, pulverized	PJ
IN_OCTOT	Industry - Other combustion	
IO_NH3_EMISS	Other industrial NH_3 emissions	kt NH3
LEAD_GASOL	Heavy and light duty vehicles: leaded gasoline (exhaust)	PJ
MINE_BC	Mining: Brown coal	Mt
MINE_HC	Mining: Hard coal	Mt
MINE_OTH	Mining: Bauxite, copper, iron ore, zinc ore, manganese ore, other	Mt
NONEN	Non-energy use of fuels	PJ
ORG_PROC	Organic chemical industry, process	kt VOC
ORG_STORE	Organic chemical industry, storage	kt VOC
OTHER_NOX	Other: (activity given as NO_x emissions in kt)	kt
OTHER_PM	Other: (activity given as PM emissions in kt)	kt
OTHER_SO2	Other: (activity given as SO_2 emissions in kt)	kt
OTH_NH3_EMISS	Other NH_3 emissions	kt NH3
PHARMA	Pharmaceutical industry	kt SLV
PIS	Products incorporating solvents	kt PG
PNIS	Products not incorporating solvents	kt VOC
PP_EX_OTH	Power & district heat plants: Exist. other	PJ
PP_EX_OTH1	Power & district heat plants: Exist. other, grate firing	PJ

SEC_ABB	NAME	INPUT_UNIT
PP_EX_OTH2	Power & district heat plants: Exist. other, fluidized bed	PJ
PP_EX_OTH3	Power & district heat plants: Exist. other, pulverized	PJ
PP_EX_WB	Power & district heat plants: Exist. wet bottom	PJ
PP_NEW	Power & district heat plants: New	PJ
PP_NEW1	Power & district heat plants: New, grate firing	PJ
PP_NEW2	Power & district heat plants: New, fluidized bed	PJ
PP_NEW3	Power & district heat plants: New, pulverized	PJ
PP_TOTAL	Power & district heat plants (total)	PJ
PRT_OFFS	Printing, offset	kt INK
PRT_OFFS_NEW	Printing, offset, new installations	kt INK
PRT_PACK	Flexography and rotogravure in packaging	kt INK
PRT_PACK_NEW	Flexography and rotogravure in packaging, new installations	kt INK
PRT_PUB	Rotogravure in publication	kt INK
PRT_PUB_NEW	Rotogravure in publication, new installations	kt INK
PRT_SCR	Screen printing	kt INK
PRT_SCR_NEW	Screen printing, new installations	kt INK
PR_ALPRIM	Ind. Process: Aluminum production - primary	Mt
PR_ALSEC	Ind. Process: Aluminum production - secondary	Mt
PR_BAOX	Ind. Process: Basic oxygen furnace	Mt
PR_BRIQ	Ind. Process: Briquettes production	Mt
PR_CAST	Ind. Process: Cast iron (grey iron foundries)	Mt
PR_CAST_F	Ind. Process: Cast iron (grey iron foundries) (fugitive)	Mt
PR_CBLACK	Ind. Process: Carbon black production	Mt
PR_CEM	Ind. Process: Cement production	Mt
PR_COKE	Ind. Process: Coke oven	Mt
PR_EARC	Ind. Process: Electric arc furnace	Mt
PR_FERT	Ind. Process: Fertilizer production	Mt
PR_GLASS	Ind. Process: Glass production (flat, blown, container glass)	Mt
PR_HEARTH	Ind. Process: Open hearth furnace	Mt
PR_HMTRA	Ind. Process: Hot metal transport in iron and steel plant	
PR_LIME	Ind. Process: Lime production	Mt
PR_NIAC	Ind. Process: Nitric acid	Mt
PR_OTHER	Ind. Process: Production of glass fiber, gypsum, PVC, other	Mt
PR_OT_NFME	Ind. Process: Other non-ferrous metals prod. - primary and secondary	Mt
PR_PELL	Ind. Process: Agglomeration plant - pellets	Mt
PR_PIGI	Ind. Process: Pig iron, blast furnace	Mt
PR_PIGI_F	Ind. Process: Pig iron, blast furnace (fugitive)	Mt
PR_PULP	Ind. Process: Paper pulp mills	Mt
PR_REF	Ind. Process: Petroleum refineries	Mt

SEC_ABB	NAME	INPUT_UNIT
PR_SINT	Ind. Process: Agglomeration plant - sinter	Mt
PR_SINT_F	Ind. Process: Agglomeration plant - sinter (fugitive)	Mt
PR_SMIND_F	Ind. Process: Small industrial and business facilities - fugitive	M persons
PR_SUAC	Ind. Process: Sulfuric acid	Mt
REF_PROC	Refineries - process	Mt crude
RESID	Combustion in residential and commercial sector	PJ
RES_BBQ	Residential: Meat frying, food preparation, BBQ	M persons
RES_CIGAR	Residential: Cigarette smoking	M persons
RES_FIREW	Residential: Fireworks	M persons
STH_AGR	Storage and handling: Agricultural products (crops)	Mt
STH_COAL	Storage and handling: Coal	Mt
STH_FEORE	Storage and handling: Iron ore	Mt
STH_NPK	Storage and handling: N,P,K fertilizers	Mt
STH_OTH_IN	Storage and handling: Other industrial products (cement, bauxite, coke	Mt
TRA_AIR_VOC	Air transport (LTO)	kt
TRA_OT	Other transport: rail (solid fuels), heating (stationary combustion)	PJ
TRA_OTS	Other transport: ships	
TRA_OTS_L	Other transport: ships; large vessels (exhaust)	PJ
TRA_OTS_M	Other transport: ships; medium vessels (exhaust)	PJ
TRA_OT_AGR	Other transport: agriculture (exhaust)	PJ
TRA_OT_AIR	Other transport: air traffic (LTO)	PJ
TRA_OT_CNS	Other transport: construction machinery (exhaust)	PJ
TRA_OT_INW	Other transport: inland waterways (exhaust)	PJ
TRA_OT_LB	Other transport: other off-road; 4-stroke (military, households, etc.)	PJ
TRA_OT_LD2	Other transport: off-road; 2-stroke (exhaust)	PJ
TRA_OT_RAI	Other transport: rail (exhaust)	PJ
TRA_RD	Light duty vehicles: cars, motorcycles (electric, renewable)	PJ
TRA_RDXLD4	Light duty vehicles: gasoline direct injection (GDI) (exhaust)	PJ
TRA_RD_HD	Heavy duty trucks and buses (exhaust)	PJ
TRA_RD_LD2	Motorcycles: 2-stroke; mopeds (also cars) (exhaust)	PJ
TRA_RD_LD4	Light duty vehicles: 4-stroke (excl. GDI) (exhaust)	PJ
TRA_RD_LF2	Transport road - 2 stroke engines	PJ
TRA_RD_M4	Motorcycles: 4-stroke (exhaust)	PJ
TRB_OT_RAI	Other transport: rail (non-exhaust)	bln btkm
TRB_RDXLD4	Light duty vehicles: gasoline direct injection (GDI) (brake wear)	bln km
TRB_RD_HD	Heavy duty trucks and buses (brake wear)	bln km
TRB_RD_LD2	Motorcycles: 2-stroke; mopeds (also cars) (brake wear)	bln km
TRB_RD_LD4	Light duty vehicles: 4-stroke (excl. GDI) (brake wear)	bln km
TRB_RD_M4	Motorcycles: 4-stroke (brake wear)	bln km

SEC_ABB	NAME	INPUT_UNIT
TRD_RDXLD4	Light duty vehicles: gasoline direct injection (GDI) (abrasion)	bln km
TRD_RD_HD	Heavy duty trucks and buses (abrasion)	bln km
TRD_RD_LD2	Motorcycles: 2-stroke; mopeds (also cars) (abrasion)	bln km
TRD_RD_LD4	Light duty vehicles: 4-stroke (excl. GDI) (abrasion)	bln km
TRD_RD_M4	Motorcycles: 4-stroke (abrasion)	bln km
TRT_RDXLD4	Light duty vehicles: gasoline direct injection (GDI) (tyre wear)	bln km
TRT_RD_HD	Heavy duty trucks and buses (tyre wear)	bln km
TRT_RD_LD2	Motorcycles: 2-stroke; mopeds (also cars) (tyre wear)	bln km
TRT_RD_LD4	Light duty vehicles: 4-stroke (excl. GDI) (tyre wear)	bln km
TRT_RD_M4	Motorcycles: 4-stroke (tyre wear)	bln km
VEHR_P	Vehicle refinishing	kt paint
VEHR_P_NEW	Vehicle refinishing (new installations)	kt paint
VEHTR	Treatment of vehicles	mln POP
WASTE_AGR	Waste: Agricultural waste burning	Mt
WASTE_FLR	Waste: Flaring in gas and oil industry	PJ
WASTE_RES	Waste: Open burning of residential waste	Mt
WASTE_VOC	Waste treatment and disposal	kt VOC
WOOD	Preservation of wood	kt SLV
WOOD_NEW	Preservation of wood (new installations)	kt SLV
WT_NH3_EMISS	Waste treatment and disposal	kt NH3

Table 3.5: List of activities in the RAINS model

ACT_ABB	NAME
BC1	Brown coal/lignite, high grade
BC2	Brown coal/lignite, low grade
CRU	Crude oil
DC	Derived coal (coke, briquettes)
DL	Dairy cows - liquid (slurry) systems
DS	Dairy cows - solid systems
ELE	Electricity
EMI	Emissions of NMVOC
ETH	Ethanol
FU	Fur animals
GAS	Natural gas (incl. other gases)
GSL	Gasoline
H2	Hydrogen
HC1	Hard coal, high quality
HC2	Hard coal, medium quality
HC3	Hard coal, low quality
HF	Heavy fuel oil
HO	Horses
HT	Heat (steam, hot water)
HYD	Hydro
INK	Printing inks
LFL	Leaded gasoline
LH	Laying hens
LPG	Liquefied petroleum gas
MD	Medium distillates (diesel, light fuel oil)
MTH	Methanol
NOF	No fuel use
NUC	Nuclear
OL	Other cattle - liquid (slurry) systems
OP	Other poultry
OS	Other cattle - solid systems
OS1	Other solid-low S (biomass, waste, wood)
OS2	Other solid-high S (incl. high S waste)
PG	Paint and glue produced
PL	Pigs - liquid (slurry) systems
PNT	Paint use
POP	Population
PS	Pigs - solid systems
REN	Renewable (solar, wind, small hydro)
SH	Sheep and goats
SLV	Solvent use
TEX	Textiles (clothing)
VEH	Vehicles

4 Modelling of emission control potentials and costs

4.1 The objectives of emission and control cost calculations within the framework of an integrated assessment model

One of the central objectives of integrated assessment models is to assist in the cost-effective allocation of emission reduction measures across various pollutants, several countries and different economic sectors. Obviously, this task requires consistent information about the costs of emission control at the individual sources, and it is the central objective of this cost module to provide such information.

The optimal allocation of emission control measures between countries is crucially influenced by differences in emission control costs for the individual emission sources. It is therefore of utmost importance to identify systematically the factors leading to differences in emission control costs among countries, economic sectors and pollutants. Such differences are usually caused, *inter alia,* by variations in the composition of the various emission sources, the state of technological development and the extent to which emission control measures are already applied.

4.2 Emission control options

There exist a large variety of options to reduce emissions from the various sources. In principle, such options can be grouped into

- behavioural changes that reduce the anthropogenic driving forces leading to emissions of pollutants. Such changes in human activities can be autonomous (e.g., changes in preferences for societal life styles), they could be fostered by command-and-control approaches (e.g., legal traffic restrictions) or they can be triggered by economic incentives (e.g., pollution taxes, emission trading systems, etc.). In the RAINS concept, such changes are reflected through alternative exogenous scenarios of the driving forces, but not internalised into the RAINS calculations.

- Structural measures that supply the same level of (energy) services but with less polluting activities. This group includes fuel substitution (e.g., switch from coal burning to natural gas) and energy conservation/energy efficiency improvements. Such measures do not modify the projection of anthropogenic driving forces, but involve far-reaching infra-structural changes with complex interactions and feedbacks within national economies. The present version of RAINS focusing on air pollution assesses the potential of such measures for emission reductions through alternative scenarios of energy consumption and agricultural activities (e.g., Syri *et al.*, 2001). The RAINS extension to greenhouse gases, which is presently under construction, will introduce such structural changes as explicit control options in RAINS and will allow their costs to be calculated.

- A large range of technical measures has been developed to capture emissions at their sources before they can enter the atmosphere. Emission reductions achieved through these options neither modify the driving forces of emissions nor change the structural composition of energy systems or agricultural activities. RAINS contains databases with a large number of pollutant-specific end-of-pipe measures and assesses their application potential and costs.

As mentioned above, the present version of RAINS restricts the endogenous analysis to end-of-pipe control measures. The cost-effectiveness of structural measures can be studied with alternative exogenous projections of the driving forces, while a comprehensive assessment of the potential of behavioural changes would require an extended perspective including social aspects.

4.3 The choice of control options for RAINS

As of now, the RAINS model restricts the internal analysis to add-on control measures. Structural changes can be evaluated through alternative driver scenarios. For the RAINS extension to greenhouse gases, however, a methodology has been developed to consider structural changes as an endogenous element of the RAINS model (Klaassen, 2004).

There exist a large number of technical measures to reduce emissions from anthropogenic activities; a full description requires considerable in-depth technical detail for all emission sources, which is hardly available even at the national level. For a pan-European assessment, it is critical to maintain the analysis at a manageable level, so that the required data can be realistically acquired from available information and can be reviewed by stakeholders with a reasonable amount of effort. Thus, RAINS groups abatement options together in a limited number of measures with comparable technical and economic features. The actual selection of abatement options is determined by pollutant-specific conditions, and is documented in the various papers describing the RAINS cost calculations (Cofala et al., 1998a; Cofala et al., 1998b; Klimont et al., 2000, Klimont et al.,, 2002).

Technical data describing the features of these options are extracted from various documents prepared for the IPPC BAT reference notes, earlier work of the UN/ECE Task Forces on Abatement Technologies and a wide body of international and national literature.

In 2001, the UN/ECE Expert Group on Techno-Economic Issues (EGTEI) started to collect information on emission control options in a systematic way, to review the information with national and industrial stakeholders and to prepare national data for direct input into RAINS. It has turned out, however, that this process could up to now produce only a limited amount of technical information, and that the full participation of national experts is hampered by the considerable complexity and the amount of data demanded by the approach. Up to now, RAINS has introduced information from EGTEI on mobile road transport, off-road sources, the glass industry and solvent use.

4.4 Cost calculation

The basic intention of a cost evaluation in the RAINS model is to identify the values to society of the resources diverted in order to reduce emissions in Europe. In practice, these values are approximated by estimating costs at the production level rather than prices to the consumers. Therefore, any mark-ups charged over production costs by manufacturers or dealers do not represent actual resource use and are ignored. Certainly, there will be transfers of money with impacts on the distribution of income or on the competitiveness of the market, but these should be removed from a consideration of the efficiency of a resource. Any taxes added to production costs are similarly ignored as transfers.

The central assumption for the RAINS cost calculation is the existence of a free market for abatement equipment throughout Europe that is accessible to all countries at the same conditions. Thus, the capital investments for a certain technology can be specified as being independent of the country. Simultaneously, the calculation routine takes into account several country-specific parameters that characterize the situation in a given region. For instance, those parameters include: average boiler sizes, capacity/vehicles utilization rates, emission factors etc.

The expenditures on emission controls are differentiated into:

- investments,
- fixed operating costs, and
- variable operating costs.

From these three components RAINS calculates annual costs per unit of activity level. Next, these costs are related to ton of pollutant abated.

Some of the parameters are considered common for all countries. These include technology-specific data, such as removal efficiencies, unit investment costs, fixed operation and maintenance costs, as well as parameters used for calculating variable cost components like extra demand for labour, energy, and materials.

Country-specific parameters characterize more closely the type of capacity operated in a given country and its operation regime. To these parameters belong: average size of installation in a given sector, plant factors, annual fuel consumption and/or mileage for vehicles. In addition, the prices for labour, electricity, fuel and other materials as well as the cost of waste disposal also belong to that category.

The following sections introduce the cost calculation principles used in RAINS and explain the construction of the cost curves that will be further used in the optimisation module of the RAINS model. Values of all parameters used to calculate country-specific costs and the national cost curves are provided on the RAINS web site (http://www.iiasa.ac.at/web-apps/tap/RainsWeb/RainsLogin.htm).

Although based on the same principles, the details of cost calculations for individual sectors differ. Thus, the formulas used for stationary combustion sources, the so-called industrial process sources and mobile sources (vehicles) are discussed separately below.

4.5 Costs for stationary combustion sources

4.5.1 Investments

Investments cover the expenditure accumulated until the start-up of an abatement technology. These costs include, e.g., delivery of the installation, construction, civil works, ducting, engineering and consulting, licence fees, land requirement and capital. The RAINS model uses investment functions where these cost components are aggregated into one function. For stationary combustion sources, the investment costs for individual control installations depend on flue gas volume treated. This in turn can be related to the boiler size *bs*. The form of the function is described by its coefficients ci^f and ci^v. Coefficients *ci* are valid for hard coal fired boilers. Thus, coefficient *v* is used to account for the different flue gas volume to be handled when other fuel is used. Additional investments, in the case of retrofitting existing boilers/furnaces, are taken into account by the retrofitting cost factor *r*. The shape of this investment function is given in Equation 4.1:

$$I = (ci^f + \frac{ci^v}{bs}) * v * (1 + r) \qquad (4.1)$$

Coefficients *ci* are estimated based on investment functions presented in Rentz et al., 1996. The original investment functions relate capital investments in Euro/1000 m^3 flue gases/h to the volume of flue gases treated (in 1000 m^3/h). These functions have been converted to a function that uses boiler size (in MW_{th}). Parameters of the function are different for three capacity classes: less than 5 MW_{th}, from 5 to 50 MW_{th} and above 50 MW_{th}.

Investments are annualised over the technical lifetime of the plant *lt* by using the real interest rate *q* (as %/100):

$$I^{an} = I * \frac{(1+q)^{lt} * q}{(1+q)^{lt} - 1} \tag{4.2}$$

4.5.2 Operating costs

The annual **fixed expenditures** OM^{fix} cover the costs of repairs, maintenance and administrative overhead. These cost items are not related to the actual use of the plant. As a rough estimate for annual fixed expenditures, a standard percentage *f* of the total investments is used:

$$OM^{fix} = I * f \tag{4.3}$$

In turn, the **variable operating costs** OM^{var} are related to the actual operation of the plant and take into account:

- additional labour demand,
- increased energy demand for operating the device (e.g., for the fans and pumps), and
- waste disposal.

These cost items are calculated with the specific demand λ^x of a certain control technology and its (country-specific) price c^x.

$$OM^{var} = \lambda^l c^l / pf + \lambda^e c^e + ef \ * \eta \ * \lambda^d c^d \tag{4.4}$$

where

η	emission removal efficiency,
λ^l	labour demand (per thermal capacity unit),
λ^e	additional electricity demand (per unit of fuel used),
λ^d	demand for waste disposal (per unit of dust reduced),
c^l	labour cost,
c^e	electricity price,
c^d	waste disposal cost,
pf	plant factor (annual operating hours at full load),
ef	unabated emission factor

4.5.3 Unit reduction costs

Unit costs per PJ fuel used

Based on the above-mentioned cost items, the unit costs for the removal of emissions can be calculated. In Equation 4.5, all the expenditures of a control technology are related to one unit of fuel input (in PJ). The investment-related costs are converted to fuel input by applying the capacity utilization factor *pf* (operating hours/year):

$$c_{PJ} = \frac{I^{an} + OM^{fix}}{pf} + OM^{var} \tag{4.5}$$

Unit costs per ton of pollutant removed

The cost effectiveness of different control options can only be evaluated by relating the abatement costs to the amount of reduced emissions. For this purpose Equation 4.6 is used:

$$c_{poll} = c_{PJ} \,/\, (\, ef_k * \eta_k \,) \tag{4.6}$$

4.6 Costs for industrial process emission sources

4.6.1 Investments

For process sources, the investment costs are related to the activity unit of a given process. For the majority of processes these are annual tons produced. For refineries, the investment function is related to one ton of raw oil input to the refinery. The investment function and annualised investments are given by Equations 4.7 and 4.8:

$$I = ci^{f} * (1 + r) \tag{4.7}$$

$$I^{an} = I * \frac{(1+q)^{lt} * q}{(1+q)^{lt} - 1} \tag{4.8}$$

4.6.2 Operating costs

The operating costs are calculated with formulas similar to those used for stationary combustion. However, since the activity unit is different the formulas have a slightly different form:

$$OM^{fix} = I * f \tag{4.9}$$

$$OM^{var} = \lambda^{l} c^{l} + \lambda^{e} c^{e} + ef \;\; * \eta \;\; * \lambda^{d} c^{d} \tag{4.10}$$

The coefficients λ^l, λ^e, and λ^d are per ton of product.

4.6.3 Unit reduction costs

Unit costs per ton of product

This cost is calculated from the following formula:

$$c_{ton} = I^{an} + OM^{fix} + OM^{var} \qquad (4.11)$$

Unit costs per ton of pollutant removed

As for combustion sources, one can calculate costs per unit of pollutant removed:

$$c_{poll} = c_{ton} / (ef_k * \eta_k) \qquad (4.12)$$

4.7 Costs for mobile sources

4.7.1 Investments

The cost evaluation for mobile sources follows the same basic approach as for stationary sources. The most important difference is that the investment costs are given **per vehicle**, not per unit of production capacity. The number of vehicles is then computed based on information on total annual fuel consumption by a given vehicle category and average fuel consumption per vehicle per year.

The following description uses the indices *i, j*, and *k* to indicate the nature of the parameters:

i	denotes the country,
j	the transport (sub)sector/vehicle category,
k	the control technology.

The costs of applying control devices to the transport sources include:

- additional investment costs;
- increase in maintenance costs expressed as a percentage of total investments; and
- change in fuel cost resulting from the inclusion of emission control.

The investment costs $I_{i,j,k}$ are given in €/vehicle and are available separately for each technology and vehicle category. They are **annualised** using Equation 4.13:

$$I^{an}_{i,j,k} = I_{j,k} \cdot \frac{(1+q)^{lt_{i,j,k}} \cdot q}{(1+q)^{lt_{i,j,k}} - 1} \qquad (4.13)$$

where:

$lt_{i,j,k}$	lifetime of control equipment.

4.7.2 Operating costs

The increase in maintenance costs (**fixed costs**) is expressed as a percentage *f* of total investments:

$$OM^{fix}_{i,j,k} = I_{i,j,k} \cdot f_k \qquad (4.14)$$

The change in fuel cost is caused by:

- change in fuel quality required by a given stage of control[1]
- change in fuel consumption after inclusion of controls

It can be calculated as follows:

$$OM^{e}_{i,j,k}(t) = \Delta c^{e}_{j} + \lambda^{e}_{j,k} * (c^{e}_{i,j} + \Delta c^{e}_{j}) \quad (4.15)$$

where:

$\lambda^{e}_{j,k}$	percentage change in fuel consumption by vehicle type j caused by implementation of control measure k,
$c^{e}_{i,j}$	fuel price (net of taxes) in country i and sector j in the base year,
Δc^{e}_{j}	change in fuel cost caused by the change in fuel quality,

This change in fuel cost is related to one unit of fuel used by a given vehicle category.

Annual fuel consumption per vehicle is a function of the consumption in the base year (t_0=1990), **fuel efficiency improvement,** and **change in activity per vehicle** (i.e., change in annual kilometers driven) relative to the base year:

$$fuel_{i,j}(t) = fuel_{i,j}(t_0) * fe_{i,j}(t) * \Delta ac_{i,j}(t) \quad (4.16)$$

where

$fe_{i,j}(t)$ - fuel efficiency improvement in time step t relative to the base year (1990 = 1)

$\Delta ac_{i,j}(t)$ - change in activity per vehicle in time step t relative to the base year (1990 = 1)

4.7.3 Unit reduction costs

The unit costs of abatement ce_{PJ} (related to one unit of fuel input) add up to

$$ce_{PJ,i,j,k}(t) = \frac{I^{an}_{i,j,k} + OM^{fix}_{i,j,k}}{fuel_{i,j}(t)} + OM^{e}_{i,j,k}(t) \quad (4.17)$$

These costs can be related to the emission reductions achieved. The costs per unit of abated emissions are as follows:

$$cn_{i,j,k}(t) = \frac{ce_{i,j,k}(t)}{ef_{i,j,k} * \eta_{j,k}} \quad (4.18)$$

The most important factors leading to differences among countries in unit abatement costs are: different annual energy consumption per vehicle and country-specific unabated emission factors. The latter difference is caused by different compositions of the vehicle fleet as well as differences in

[1] This cost component takes into account higher fuel price caused by the change in fuel specification (e.g., different contents of aromatics or benzene, different cetane number)

driving patterns (e.g., different share of urban vs. highway driving depending on available infrastructure in a given country).

4.8 Marginal reduction costs

Marginal costs relate the extra costs for an additional measure to the extra abatement of that measure (compared to the abatement of the less effective option). RAINS uses the concept of marginal costs for ranking the available abatement options, according to their cost effectiveness, into the so-called “national cost curves”.

If, for a given emission source (category), a number of control options *M* are available, the marginal costs mc_m for control option *m* are calculated as

$$mc_m = \frac{c_m \eta_{l\,m} - c_{m-1} \eta_{l\,m-1}}{\eta_{l\,m} - \eta_{l\,m-1}} \tag{4.19}$$

where

c_m	unit costs for option *m* and
η_{lm}	pollutant *l* removal efficiency of option *m (l= pollutant)*

4.9 Constructing a cost curve

For each emission scenario RAINS creates a so-called emission reduction cost curve. Such cost curves define - for each country and year - the potential for further emission reductions beyond a selected initial level of control and provide the minimum costs of achieving such reductions. For a given abatement level, a cost-optimal combination of abatement measures is defined.

In the optimisation module of RAINS, cost curves capturing the remaining measures beyond the baseline scenario are used to derive the internationally cost-optimal allocation of emission reductions to achieve pre-selected environmental targets (e.g., desired human health or ecosystems protection level).

Cost curves are compiled by ranking available emission control options for various emission sources according to their cost-effectiveness and combining them with the potential for emission reductions determined by the properties of sources and abatement technologies. Based on the calculated unit cost, the cost curve is constructed first for every sector and then for the whole region (country), employing the principle that technologies characterized by higher costs and lower reduction efficiencies are considered as not cost-efficient and are excluded from further analysis. The marginal costs (costs of removing an additional unit of pollutant by a given control technology) are calculated for each sector. The remaining abatement options are finally ordered according to increasing marginal costs to form the cost curve for the country being considered.

RAINS computes two types of cost curves:

- The ‘total cost’ curve displays total annual costs of achieving certain emission levels in a country. These curves are piece-wise linear, with the slopes for individual segments determined by the costs of applying the various technologies.

- The ‘marginal cost’ curve is a step-function, indicating the marginal costs (i.e., the costs for reducing the last unit of emissions) at various reduction levels. The algorithm for calculating the marginal costs is explained in Section 4.8.

The cost curve can be displayed in RAINS in tabular or graphical form. Each curve concerns a selected country (or region of a country), emission scenario and year. The table includes columns listing activity type (e.g. fuel combustion), economic sector, control technology combinations, marginal costs (in €/ton pollutant removed), remaining emissions (i.e., initial emission less cumulative emissions removed, in kt), and total cumulative control costs in million €/year.

An example of a cost curve is presented in Table 4.1. The first row in the table shows initial emissions for a given year and in a given country The amount of particulate matter reduced by a particular technology can be derived by comparing the emissions given for this option in the column *“Remaining emissions”* with the preceding value. The *"Total cost"* column displays cumulative costs. This means that for any emission level a cost value in this column represents total costs incurred to achieve this level of emissions. The examples presented in these tables contain only part of a cost curve, which typically includes up to 300 control options ordered according to increasing marginal costs (such a complete cost curve is presented in Figure 4.2).

A graphical representation of Table 4.1 is presented in Figure.4.1. The remaining emissions of TSP are on the x-axis and the total cost on the y-axis. The highest emission value is called the initial emissions and the lowest level is often referred to as maximum feasible reduction (MFR). In the literature, cost curves are often presented in different ways such that instead of showing remaining emissions, the amount of pollutant reduced is shown on the x-axis. As can be seen, the abatement achieved, as well as the cost involved, varies substantially from technology to technology. Note the marked points that indicate the technologies appearing in the same order as in Table 4.1.

Table 4.1: Example of a no-control cost curve for TSP (only part of it).

Activity code	Sector code	Technology code	Marginal cost €/t TSP	Remaining emissions 10^6 tons	Total cost 10^6 €/a
	Initial emissions			15.07	0.0
NOF	PR_CEM	PR_CYC	2.6	12.39	7.0
NOF	PR_FERT	PR_CYC	3.4	12.29	7.3
NOF	PR_LIME	PR_CYC	7.3	11.90	10.2
NOF	PR_CEM	PR_ESP1	7.5	11.13	15.9
NOF	PR_FERT	PR_FF	9.9	11.08	16.5
NOF	PR_ALPRIM	PR_CYC	17.5	11.06	16.8
NOF	PR_EARC	PR_CYC	19.4	10.90	19.9
NOF	PR_SINT	PR_CYC	21.7	10.73	23.6
BC2	PP_NEW3	ESP1	23.3	10.18	36.5
BC2	PP_NEW2	ESP1	23.5	10.03	40.0

NOF	PR_COKE	PR_CYC	23.8	10.01	40.4
BC2	PP_EX_OTH3	ESP1	23.9	6.72	119.1
NOF	PR_ALPRIM	PR_ESP1	24.2	6.71	119.3
BC2	PP_EX_OTH2	ESP1	24.4	5.81	141.2
NOF	PR_CEM	PR_ESP2	26.4	5.70	144.2
HC2	PP_NEW3	ESP1	27.3	5.52	149.1
HC2	PP_NEW2	ESP1	27.6	5.47	150.5
HC2	IN_OC3	IN_ESP1	28.6	5.32	154.9
HC2	IN_OC2	IN_ESP1	29.0	5.21	157.9
HC2	PP_EX_OTH3	ESP1	29.2	3.03	221.7
BC2	PP_EX_OTH1	CYC	29.2	3.00	222.6
NOF	PR_COKE	PR_ESP1	30.1	2.99	222.9
HC2	PP_EX_OTH2	ESP1	30.1	2.36	241.9
HC2	IN_BO3	IN_ESP1	32.2	2.34	242.6
BC2	IN_BO3	IN_ESP1	32.5	2.32	243.0
HC2	IN_BO2	IN_ESP1	33.1	2.31	243.6
BC2	IN_BO2	IN_ESP1	34.2	2.30	243.8
BC2	PP_NEW3	ESP2	36.4	2.28	244.5
NOF	PR_EARC	PR_FF	36.5	2.18	248.1
HC2	IN_OC1	IN_CYC	38.7	2.16	249.2
….	….	….	….	….	….

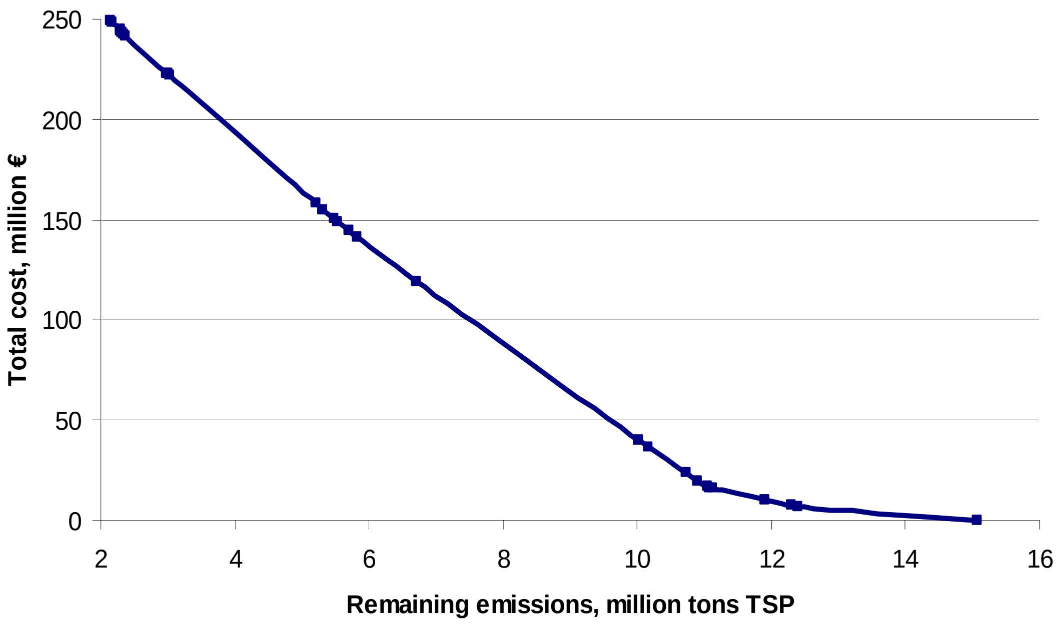

Figure.4.1: Graphical illustration of the part of the TSP cost curve presented in Table 4.1.

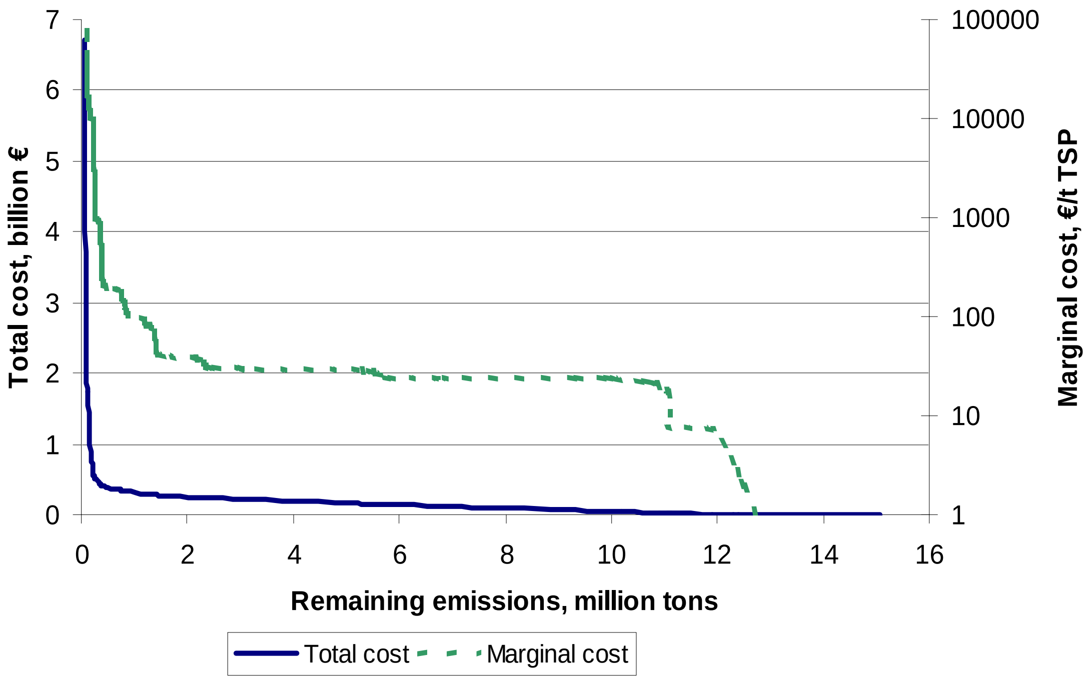

Figure 4.2: Example of the complete no-control TSP cost curve.

4.10 Validation and uncertainties

Only in a few cases can the future emission control potentials and costs estimated by the RAINS model be directly compared with real world observations. Two studies have been conducted by stakeholders that allow a validation of the RAINS estimates:

- In 2003, CONCAWE, the environmental organization of the European oil industries, compiled estimates of NO_x control costs for European refineries based on their own databases (White, 2003) .
- A study produced in the course of the preparation for the NEC Directive by AEA Technology for the UK Department on the Environment, Transport and the Regions in 1998 has developed independent estimates of cost curves for the UK (Passant *et al.*, 1998).

While neither of these studies dwells on real-world observations of abatement cost, they are valuable benchmarks to compare with the RAINS results because they applied methodologies that are consistent with the RAINS estimates and employ more detailed national and sectoral data than available in RAINS.

4.10.1 Costs of NO_x reductions in refineries

In 2003, CONCAWE estimated costs of alternative options to control NO_x emissions from European refineries as an input to the UN/ECE EGTEI process (White, 2003). This study recognizes the importance of judging the cost-effectiveness of alternative measures in terms of unit reduction costs in €/tonne NO_x removed, which requires not only the determination of costs (numerator) but also of the "uncontrolled" (raw gas) concentration of NO_x along with the removal efficiency, which allows the determination of the denominator in Equation 4.6.

The basic cost data were derived from the CONCAWE Report 99/01 "Best available techniques to reduce emissions from refineries", which has been submitted as an input to the IPPC BREF activity. Data on a range of sizes of combustion units in European refineries (key input to cost) and ranges of "uncontrolled" NO_x concentrations (key input to quantity of NO_x removed by given technology) have been derived from survey data on more than 100 combustion units. This database also provided details on type/size of the combustion units (e.g., process furnace, boiler etc), the type of burners, type/characteristics of fuel and level of air preheat in each unit.

For Low-NO_x burners, the study explored an efficiency range from 40 to 75 percent, used a lowest achievable concentration of NO_x of 60mg/Nm^3 from a range of uncontrolled NO_x concentrations from 150 to 700 mg/Nm^3. Industrial data suggest reference capital costs for a 28 MW Unit in the range 200-600 k€, zero reference operating cost, a function describing economies of scale (Cost vs Unit Size = Cost Ref * [MW/MWref]^0.8) similar to that used in RAINS and analysed the resulting costs for a capacity range from 30 to 150 MW.

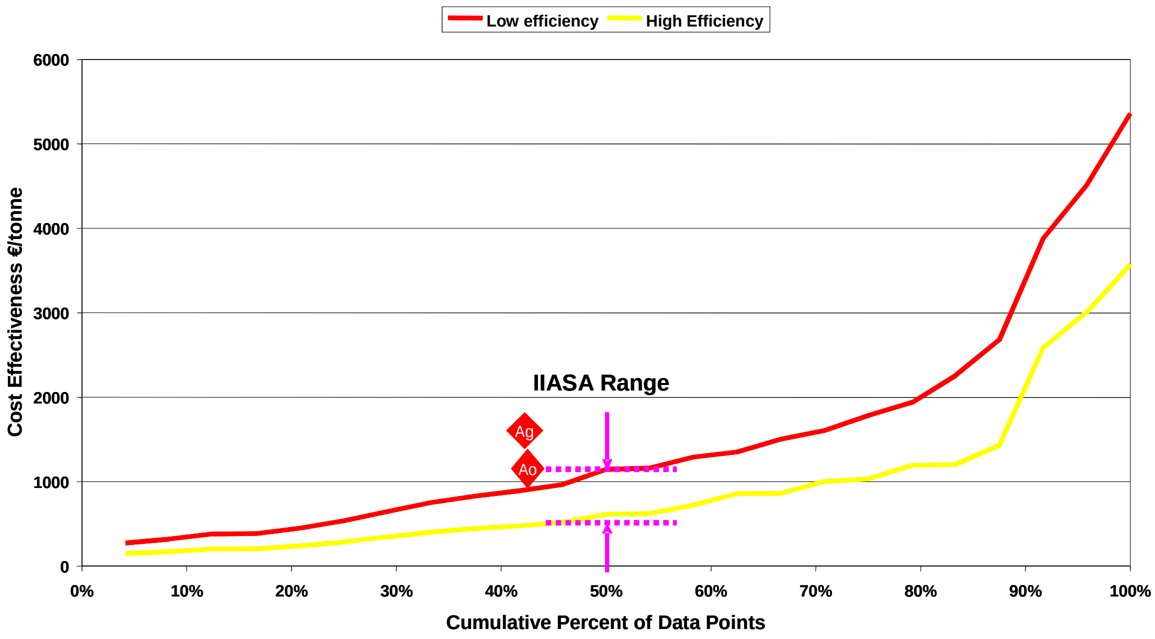

Figure 4.3: Cost-effectiveness of Low NO_x burners in refineries (in €/ton) as estimated by CONCAWE for the full range of European refineries (White, 2003) compared with the range calculated by RAINS and two estimates quoted in the EGTEI study.

For Low-NO_x burners, the RAINS estimates for the median capacity range show excellent agreement with the industry estimates (Figure 4.3). Similar conclusions emerge for Selective Non-catalytic reduction (Figure 4.4) and Selective Catalytic Reduction (Figure 4.5).

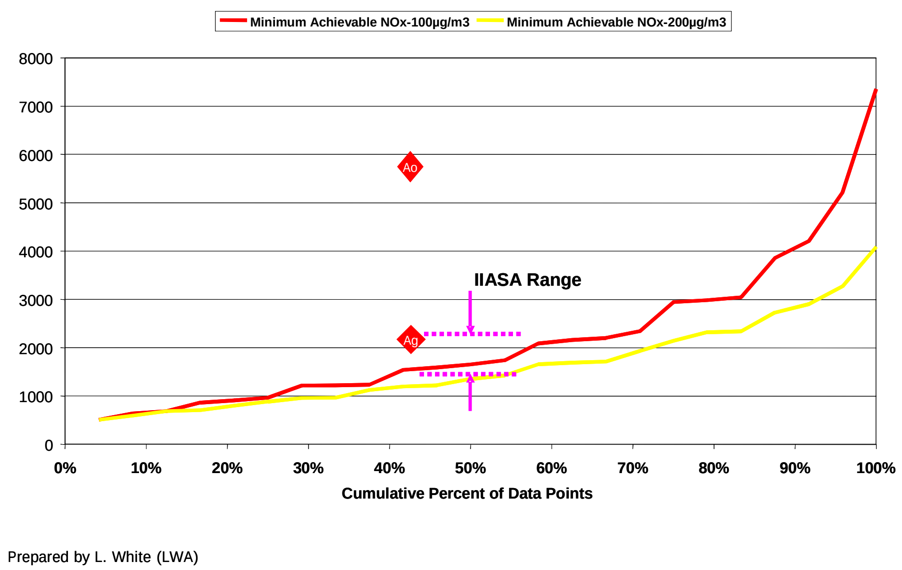

Figure 4.4: Cost-effectiveness of Selective Non-catalytic Reduction in refineries (in €/ton) as estimated by CONCAWE for the full range of European refineries (White, 2003) compared with the range calculated by RAINS and two estimates quoted in the EGTEI study.

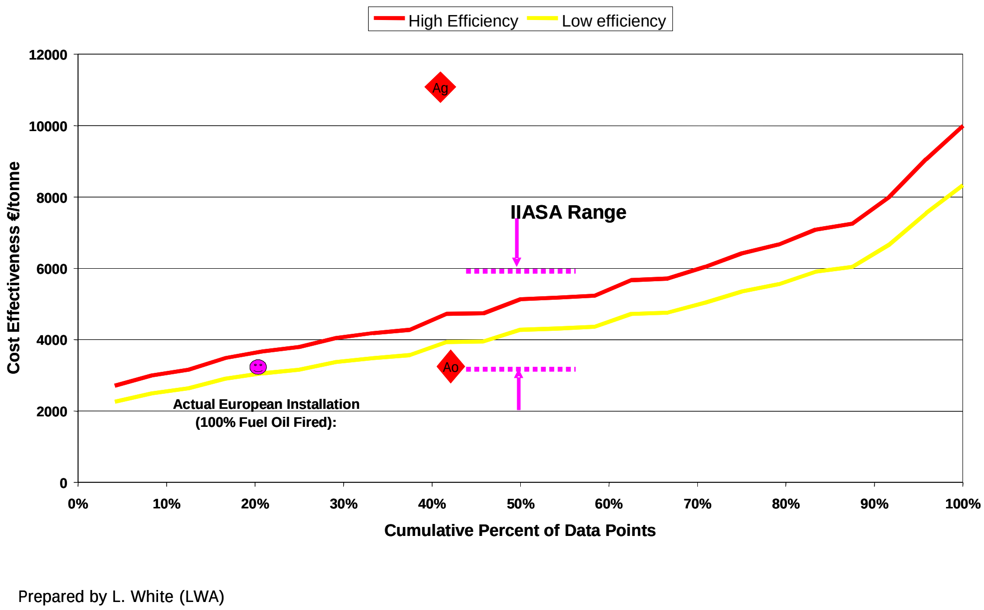

Figure 4.5: Cost-effectiveness of Selective Catalytic Reduction in refineries (in €/ton) as estimated by CONCAWE for the full range of European refineries (White, 2003) compared with the range calculated by RAINS and two estimates quoted in the EGTEI study.

4.10.2 National cost curves for UK

In the course of the review of the RAINS cost curves for the analyses of the Emission Ceilings Directive and the Gothenburg Protocol, the UK Department for the Environment, Transport and the Regions commissioned AEA Technology to compare the UK reduction potentials and costs with the estimates of the RAINS model (Passant *et al.*, 1998). The analysis was carried out for SO_2, NO_x and VOC. The analysis examined the data IIASA has put into the cost module of RAINS, tested the validity of the assumptions made, investigated whether any significant omissions exist in the input data, particularly of abatement measures important for the UK and identified critical areas where national data is required to better reflect the UK situation.

A sensitivity analysis was performed to determine the likely effect of the uncertainties on the UK cost curves (Figures 1.6 to 1.8). The RAINS cost curves are plotted against the UK mean curve; the 5 percentile and 95 percentile curves have been included to indicate the uncertainty of the estimates. The analysis concluded at this stage that the configuration of RAINS at that time generally overstated the applicability of control measures while underestimating their costs.

A particularly large discrepancy emerged for the SO_2 cost curve, where the national data suggest the maximum feasible emission controls at about 900 kilotons of SO_2, while the RAINS methodology suggested a level of approximately 250 kt to be technically achievable. The bilateral consultations between IIASA and the UK experts in 1998 helped to identify the sources of these discrepancies (mainly the applicability of emission control measures under UK conditions), and the UK eventually signed in the Gothenburg Protocol a commitment to reduce its SO_2 emissions in 2010 to a level of 585 kt.

For NO_x, the RAINS cost curve was - over a wide range - within the uncertainty band identified by AEAT, while for VOC the initial national cost curve estimated about 1200 kt as the best that could be achieved with technical measures, while the original RAINS cost curve suggested approximately 800 kt as achievable. In the Gothenburg Protocol, the UK accepted the legal obligation to reduce its VOC emissions to a level of 1200 kt.

Two aspects should be highlighted from this analysis: First, there is a tendency in many national analyses to underestimate the potentials for further emission reductions. In many cases, the assessment of further potentials is strongly influenced by current practices, operating experience and potentially by domestic political interests of stakeholders, and there seems to be limited international exchange of such practical experience from more progressive countries to others. Second, at least the national estimate from the UK, which includes more details and presumably more options than the RAINS model, considers the cost estimates of RAINS to be on the low side. This assertion is in contrast to other claims that suggest serious overestimates of RAINS abatement cost estimates due to the exclusion of structural changes in energy systems and energy conservation.

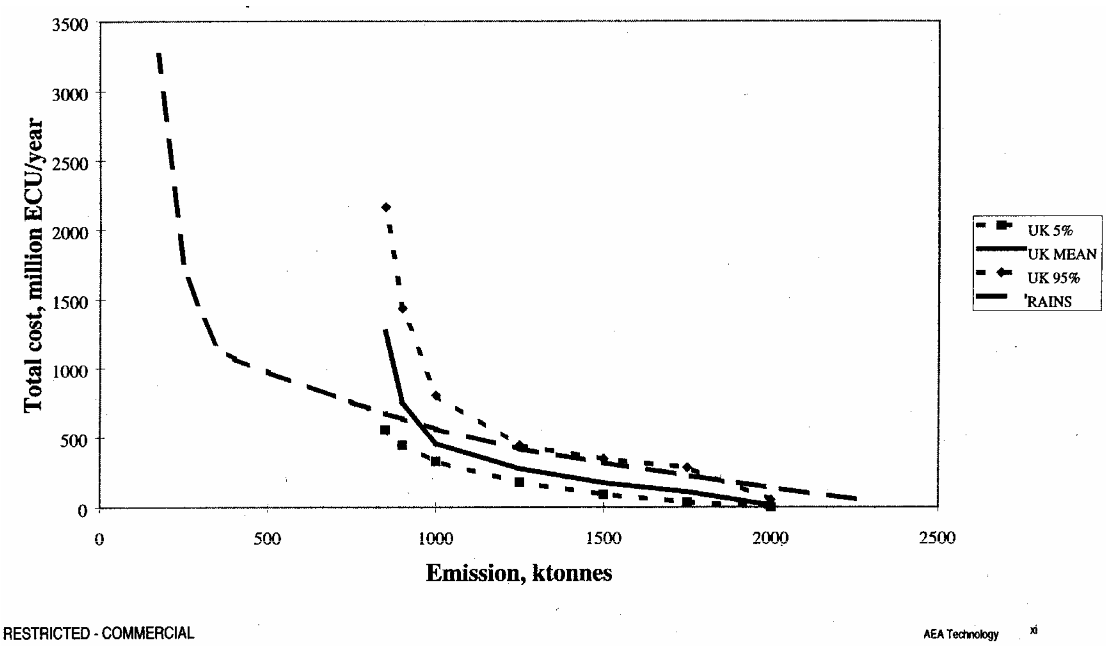

Figure 4.6: Comparison of SO_2 cost curves for the UK for the year 2010: national estimates with uncertainty ranges and RAINS. Source: Passant *et al.*, 1998

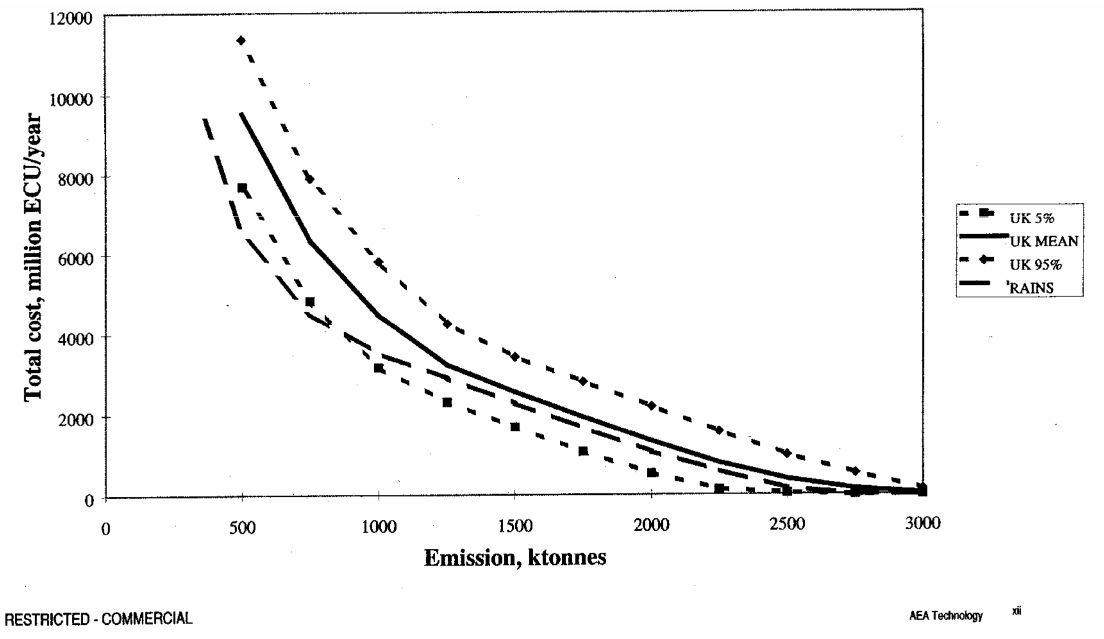

Figure 4.7: Comparison of NO_x cost curves for the UK for the year 2010: national estimates with uncertainty ranges and RAINS. Source: Passant *et al.*, 1998

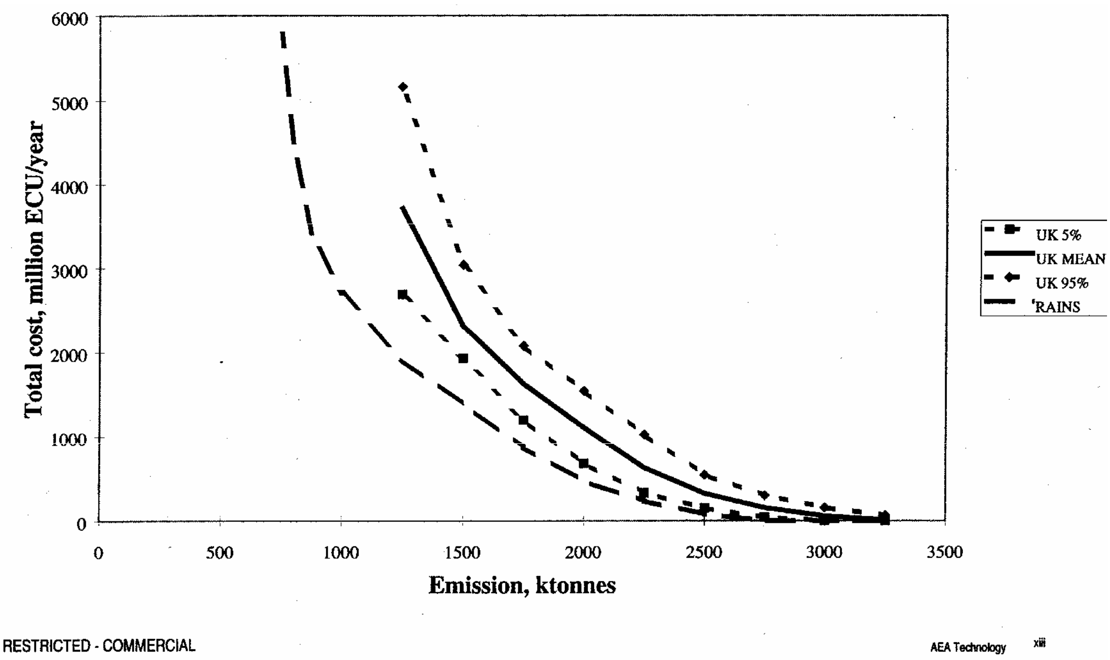

Figure 4.8: Comparison of VOC cost curves for the UK for the year 2010: national estimates with uncertainty ranges and RAINS. Source: Passant *et al.*, 1998

4.11 References

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5 Modelling of health impacts of fine particles

5.1 Modelling health impacts from fine particles

Over the past decade epidemiological studies in Europe and worldwide have measured increases in mortality and morbidity associated with air pollution. Studies in the United States have shown that those living in less polluted cities live longer than those living in more polluted cities (Dockery *et al.*, 1993; Pope *et al.*, 1995). After adjustments for other factors, an association remained between ambient concentrations of fine particles and shorter life expectancy. These findings were confirmed by a reanalysis of the original studies published by the Health Effects Institute (Krewski *et al.*, 2000) and by a recently published large-scale assessment of mortality based on data collected by the American Cancer Society (Pope *et al.*, 2002).

With accumulating evidence about health effects of air pollution, interest is growing to use data from these studies to inform environmental policies. The World Health Organization (WHO) has produced a guideline document ("*Evaluation and use of epidemiologic evidence for environmental health risk assessment"),* providing a general methodology for the use of epidemiological studies for health impact assessment (WHO, 2000). In 2001, WHO convened a working group to examine several of the aspects introduced in this report as they apply specifically to air pollution health impact assessment (WHO, 2001).

Following these guidelines from WHO this report develops a methodology for estimating losses in life expectancy due to air pollution and presents an initial implementation assessing the implications of present and future policies in Europe to control exposure to particulate matter. At this point in time, the paper focuses on the methodological framework in order to demonstrate how information relevant for health impact assessment can be put together in a consistent and meaningful way. It integrates population data, findings from epidemiological studies, information about the formation and dispersion of fine particles in the atmosphere, estimates of present and future levels of emissions of fine particles and their precursors. Awaiting further refinements in the scientific disciplines, the quantitative implementation should be considered as preliminary and needs to be revised as soon as more substantiated scientific information becomes available.

5.2 Approach

RAINS uses the following basic steps to estimate health impacts of air pollution control scenarios:

1. Obtain, for all European countries, information (a) on current mortality rates from UN population statistics and (b) on future baseline mortality rates that are implied by the UN world population projections.
2. Estimate exposure of the European population to particulate matter pollution (a) for 1990, (b) for 2010 assuming implementation of presently decided emission controls, and (c) for the lowest PM levels that could hypothetically be achieved by full application of present-day technical emission controls. This requires (i) spatially explicit information about population densities, and (ii) spatially explicit information of PM levels resulting from the three emission scenarios.

3. Using associations between particulate matter pollution and mortality found by epidemiological studies, determine the modification of mortality rates due to PM pollution.
4. Calculate changes in life expectancy (compared to the baseline UN scenario) resulting from the modified exposures to PM pollution of the three emission scenarios.
5. Examine how sensitive these estimates are to changes in the underlying assumptions.

With this approach, the RAINS combines information about

- results from epidemiological studies that quantify mortality impacts of exposure to air pollution,
- demographic structures in the various European countries and their expected development over time,
- geographically explicit estimates of exposure to air pollution, based on gridded population data and concentration fields of fine particulate matter, distinguishing urban and rural areas,
- the formation and dispersion of aerosols (fine particles) in the atmosphere from
- primary emissions of fine particles as well as the precursor emissions (sulphur dioxide, nitrogen oxides, ammonia, volatile organic compounds) leading to secondary aerosols,
- the situation estimated for 1990, the predicted conditions in the year 2010 if presently decided emission control strategies were fully implemented and the maximum technically feasible emission controls that could be achieved in the year 2010, taking into account the presently envisaged economic development in the various European countries.

5.2.1 Endpoint: Loss in life expectancy

Exposure to outdoor air pollution is associated with a broad spectrum of acute and chronic health effects ranging from irritant effects to death (American Thoracic Society (ATS), 1996a,b). While all these outcomes are potentially relevant for health impact assessment, this study restricts itself to the quantification of changes in mortality resulting from alternative air pollution control scenarios.

Associations between air pollution exposure and mortality have been assessed through two types of epidemiological studies:

- Time series studies of daily mortality measure the proportional increase in the daily death rate attributable to recent exposure to air pollution.
- Cohort studies follow large populations for years and relate their mortality to their exposure to air pollution over extended periods.

Both designs provide estimates of relative risk of mortality that can be associated with exposure to air pollution. It is important to point out that the relative risks derived from time series and cohort studies have different meanings, but refer to similar effects of air pollution: in both cases, pollution-related mortality reflects a combination of acute and chronic effects (Englert, 1999).

The WHO working group on health impact assessment (WHO, 2001) concluded that both designs could contribute useful, albeit different, information. Through their design, time series studies yield estimates of “premature” deaths due to recent exposure, in all likelihood among those who are frail due to either chronic disease, or to some transient condition. Because such studies cannot quantify chronic effects of long-term exposure, some deaths attributable to air pollution will be missed and the extent to which air pollution advances the time of death cannot be quantified (Kuenzli, 2001; McMichael, 1998). For this reason, the use of risk estimates from time series studies of daily mortality will in most cases underestimate the impact of pollution exposure on both the attributable numbers of deaths and average lifespan in a given population.

Therefore, the WHO working group on health impact assessment (WHO, 2001) concluded that the most complete estimate of both attributable numbers of death and average reduction in lifespan associated with the exposure to air pollution are those based on cohort studies. Such studies include not only those whose deaths were advanced by recent exposure to air pollution, but also those who died from chronic disease caused by long-term exposure.

The arguments of the Working Group have been further substantiated by Rabl, 2003, showing that the number of deaths is not meaningful for air pollution, whereas loss of life expectancy (LLE) is an appropriate impact indicator. The usual short-term (time series) studies yield a change in daily number of deaths attributable to acute effects of pollution, without any information on the associated LLE (although some information on this has recently become available by extending the observation window of time series). Long-term studies yield a change in age-specific mortality which makes it possible to calculate the total population averaged LLE (acute and chronic effects), but not the total number of premature deaths attributable to air pollution. The latter is unobservable because one cannot distinguish whether few individuals suffer a large or many a small LLE.

In its review of the RAINS methodology, the UN/ECE-WHO Task Force on Health (TFH, 2003) "*agreed that both the reduction in life expectancy and the total number of years of life lost were relevant informative end points to be used in the scenario analysis*."

5.2.2 Review of cohort studies

Due to the complexity of conducting cohort studies, only few analyses are available that examine the relation between long-term exposure to air pollution and mortality. These studies quantify relative risks (RR) of mortality that can be attributed to changes in exposure to air pollution. Table 5.2 summarizes these studies.

An early attempt was made in 1991 by Abbey *et al.*, to look for relationships between air pollution and mortality using health data of Californian Seventh-Day Adventists communities. At that time, statistical analysis was hampered by the non-availability of measurements of fine particulate matter (PM2.5), so that only relations with total suspended particles (TSP) could be examined. No consistent associations between TSP and mortality were found. The study was updated in 1999, following 6,338 subjects from 1977 to 1992 and extending it to PM10 (Abbey *et al.*, 1999). After corrections for age, past smoking, education, occupation and body mass index, a positive association between all-cause mortality and the number of days with PM10 above 100 $\mu g/m^3$ was found for males, but not for females. No associations were found with mean PM10, nor with cardiopulmonary or respiratory mortality.

In 1993, Dockery *et al.* analysed the mortality of 8000 adults living in six cities in the USA. This "Six Cities Study" followed cohorts of adults aged 25-74 over 14-16 years. The study estimated a relative risk (RR) of 1.14 for a 10 $\mu g/m^3$ increase in PM10, which corresponds to an 11% change in mortality for each 10 μg/m^3 change in PM2.5. The 95 percent confidence interval of RR was determined at 1.04-1.24.

The largest study using data of the American Cancer Society (ACS) examined the linkage between air pollution and mortality for more than 500,000 people aged older than 30 years in the USA over a time period of eight years (Pope *et al.*, 1995). For fine particulate matter (PM2.5), a relative risk of 1.07 for all-cause mortality (equivalent to a 6.8 percent change in mortality per 10 μg PM2.5/m^3) was found. The 95 percent confidence interval of RR was estimated at 1.04 to 1.11.

In the year 2000, the Health Effects Institute (Krewski *et al.*, 2000) conducted a reanalysis of the original Six City (Dockery *et al.*, 1993) and ACS (Pope *et al.*, 1995) cohort studies. This reanalysis assured the quality of the original data, replicated the original results, and tested those results against alternative risk models and analytic approaches without substantively altering the original findings of an association between indicators of particulate matter air pollution and mortality. In particular, it reconfirmed the relative risks found in the original studies for associations with PM2.5. Smaller associations with mortality were shown for PM15 and PM15-2.5 (coarse particles).

A recent study (Pope *et al.*, 2002) extended the time span of the ACS study to 16 years and tested possible associations of mortality with a wide range of explanatory variables (age, sex, race, smoking, education, marital status, body mass, alcohol consumption, occupational exposure and diet). It was found that fine particulate (PM2.5) and sulphur oxide pollutants were associated with all-cause, lung cancer and cardiopulmonary mortality (Table 5.1). Using the Cox proportional hazard model, the study conducted separate analyses for PM observations of the period (1979-1983) of the first ACS study, for the follow-up period (1999-2000) and for both periods combined.

Table 5.1: Adjusted mortality relative risks (RR) associated with a 10 μg/m^3 change in PM2.5 (Source: Pope *et al.*, 2002).

	Adjusted RR (95% confidence interval)		
Cause of mortality	1979-1983	1999-2000	Average
All-cause	1.04 (1.01-1.08)	1.06 (1.02-1.10)	1.06 (1.02-1.11)
Cardiopulmonary	1.06 (1.02-1.10)	1.08 (1.02-1.14)	1.09 (1.03-1.16)
Lung cancer	1.08 (1.01-1.16)	1.13 (1.04-1.22)	1.14 (1.04-1.23)
All other causes	1.01 (0.97-1.05)	1.01 (0.97-1.06)	1.01 (0.95-1.06)

Consistent associations were found between ambient levels of PM2.5 and all-cause mortality, cardiopulmonary mortality and lung cancer. For the first period, the relative risks were found to be slightly smaller than those determined in the original study, while the RR resulting from the extension up to the year 2000 match the original estimates. Measures of coarse particle fraction and total suspended particles were not consistently associated with mortality.

A Dutch cohort study (the only European study) used different metrics (BS and NO_2 as proxy for combustion-related pollution), to circumvent the problem of rather uniform PMx levels and poor availability of historical PM2.5 measurements in the Netherlands. A random sample of 5000 people was followed in this cohort study (Hoek *et al*., 2002). The association between exposure to air pollution and (cause-specific) mortality was assessed with adjustment for potential confounders. Cardiopulmonary mortality was associated with living near a major road (relative risk 1.95, 95% CI 1.09-3.52) and, less consistently, with the estimated ambient background concentration (1.34, 0.68-2.64). The relative risk for living near a major road was 1.41 (0.94-2.12) for total deaths. Non-cardiopulmonary, non-lung cancer deaths were unrelated to air pollution (1.03, 0.54-1.96 for living near a major road). Even though different metrics were used to characterise air pollution, and though the Dutch study assigned individual estimates of ambient air pollution, the authors conclude that their results are in line with the findings of the Six City study and the ACS study, showing that long-term exposure to (traffic-related) air pollution may shorten life expectancy.

Table 5.2: Available cohort studies

Study	Study object	Relative risk (RR) for all-cause mortality
Abbey *et al.*, 1991 (Seventh-Day Adventists study)	TSP 6303 non-smoking Seventh-Day Adventists in California from 1977-1986 Al-cause mortality	No correlation between TSP and all-cause mortality found
Abbey *et al.*, 1999 Update of Seventh-Day Adventists study	PM10 6338 non-smoking Seventh-Day Adventists in California from 1977-1992 All-cause mortality	RR=1.12 (1.01-1.24) for 10 $\mu g/m^3$ PM10
Dockery *et al.*, 1993 (Six Cities Study)	PM 2.5 8000 adults in 6 cities in USA followed up for 14-16 years from 1974-1991, Age: 25-74 at enrolment (max. 90 at end) All-cause mortality	RR=1.13 (1.04-1.24)
Pope *et al.*, 1995 (American Cancer Society, ACS Study)	PM 2.5 Cohort of >552,138 living in 151 cities in US for 7 years from 1982-1989 Age: 30+ at enrolment Average annual all-cause mortality	RR=1.07 (1.04-1.11)
Krewski *et al.*, 2000 (HEI Re-analysis)	PM2.5 Re-analysis of Pope *et al*. (1995) and Dockery *et al*. (1993)	Re-analysis of Dockery *et al.* : RR=1.14 Pope *et al.* (1995): RR=1.07
Pope *et al.*, 2002	PM2.5 Analysis of ACS data for 116 cities in the US for 16 years Age: 30+ at enrolment All-cause mortality, cardiopulmonary mortality, lung cancer	For 1979-1983: RR=1.04 (1.01-1.08) For 1999-2000: RR=1.06 (1.02-1.14) For 1979-2000: RR=1.06 (1.02-1.11)

The choice of PM2.5 as an indicator for PM related mortality is strongly supported by a review that was recently (spring 2003) completed by a WHO working group (WHO, 2003). The group adopted a recommendation to use fine particulate matter, ($PM_{2.5}$), as an indicator for health effects induced by particulate pollution such as increased risk of mortality in Europe.

The Task Force on Health of the United Nations Economic Commission, when conducting the in-depth review of the RAINS approach for modelling health impacts of fine particles (TFH, 2003), noted "*that some data suggested that different components that contributed to PM2.5 mass might not be equally hazardous. In particular, the discussion focused on the role of the secondary inorganic aerosols (including nitrates and sulphates). It concluded that, due to the absence of compelling toxicological data about different PM components acting in a complex mixture, it was not possible to quantify the relative importance of the main PM components for effects on human health at this stage*." Therefore, it was recommended to relate health impacts to total mass of $PM_{2.5}$, until more specific evidence becomes available.

5.2.3 Personal exposure versus cohort exposure

It is often suggested that personal exposure of individuals may not be well represented by ambient concentrations of pollutants in urban background air, which are usually monitored on a routine basis. As shown by a number of studies, the relation between personal exposure and background concentration depends on the pollutant under consideration, particularly on its dispersion characteristics and whether significant indoor pollution sources exist (e.g., gas cooking for NO_2). While for individuals such relationships were found to be weak, for larger groups of people ambient background concentrations of PM2.5 represent well the characteristic exposure (Boudeta, 2001).

Only few studies have addressed whether ambient long-term PM concentrations predict long-term personal exposure to PM well. Analyses conducted within the EXPOLIS study have suggested that long-term ambient PM concentrations predict the population average of a series of personal PM2.5 measurements well (Jantunen et al., 2002)

For purposes of health impact assessment WHO (2001) has pointed out that, while it is common to refer to the results of epidemiological studies of air pollution as providing estimates of the exposure-response relation, most epidemiological studies actually measure the relation between ambient concentration and response. Thus a health impact assessment, to be consistent with the original evidentiary studies, relates to ambient concentrations rather than to actual personal exposure.

5.2.4 The Cox Proportional Hazards Model

For estimating the concentration-response function, the epidemiological studies described above used the Cox proportional hazards model (Cox, 1972). The proportional hazards model postulates that changing the stress variable (here the change in PM concentrations) is equivalent to multiplying the hazard rate (here the mortality rate) by a proportionality factor, which is here the relative risk function. The fatalities due to PM impacts are usually assumed to be Poisson-distributed, thus the concentration-response function is of log-linear type. The Cox proportional hazard model expresses the number of fatalities in a time period as a function of the baseline fatalities and PM concentrations:

$$y = y_0 * e^{\beta * PM} \tag{1}$$

with y number fatalities

y_0 baseline fatalities

PM PM concentrations

β functional parameter

With the baseline mortality rate μ_0 defined as the quotient of baseline fatalities y_0 and population size P, the adjusted mortality rate $\overline{\mu}$ is calculated as

$$\overline{\mu} = \frac{y}{P} = \frac{y_0}{P} * e^{\beta * PM} = \mu_0 * e^{\beta * PM} . \tag{2}$$

The factor multiplying the baseline hazard rate is also termed “relative risk” *RR*, which is determined as

$$RR(PM) = e^{\beta * PM} . \tag{3}$$

In the epidemiological studies discussed above, beta is found to be low and the RR function to behave quasi-linearly in the exposure range studied (Pope *et al.*, 2002, p. 1136). Thus, RR can be approximated linearly around 0 by a first-order Taylor series in order to speed up calculations and to facilitate sensitivity und uncertainty analyses:

$$RR(PM) = (\beta . PM) + 1 . \tag{4}$$

5.2.5 Calculating life expectancy from mortality rates

Using the Cox proportional hazards model, a methodology was developed to calculate impacts of various scenarios of precursor emissions of fine particles on the life expectancy of the European population.

The methodology starts from the cohort- and country-specific mortality taken from life tables and calculates for each cohort the survival function over time. The survival function is modified by exposure to PM pollution, and can then be converted into reduced life expectancy for an individual person. The calculation uses life-tables and applies an approximation method described in Vaupel and Yashin (1985) for the calculation of the change in life expectancy. A similar approach was developed by Rabl, 2003.

For an age cohort c of age c at starting time s (here 2010) in a grid cell, the change in life expectancy can be calculated as follows:

The basis for the calculation of life expectancy is the so-called survival function $l(t)$ that indicates the percentage of a cohort alive after time t has elapsed since starting time s. $l(t)$ is an exponential function of the sum of the mortality rates $\mu_{a,b}$, which are derived from the life table with a as age and b as calendar time. As the relative risk function taken from Pope *et al.* (2002) applies only to cohorts that are at least 30 years old, younger cohorts were excluded from this analysis. Accordingly, for an age cohort aged c at start s, $l_c(t)$ is:

$$l_c(t) = e^{-\sum_{z=c}^{t} \mu_{z,z-c+s}} \tag{5}$$

where c=30, 35,...,95.

Thereby, $l_{30}(t)$ signifies the cohort of age 30 at starting time 2010, μ(30,2010) is the mortality rate for this age cohort in 2010 and μ(35,2015) the mortality rate in 2015 for the same cohort, which will be by then five years older.

The remaining life expectancy e_c for a cohort aged c is the integral from c to w_1 over $l_c(t)$:

$$e_c = \int_c^{w_1} l_c(t)\,dt \tag{6}$$

where w_1 is the maximum age considered (in this study 95 years, this age group also contains persons older than 95).

Exposure to different PM concentrations changes the mortality rate and consequently life expectancy:

$$\bar{e}_c = \int_c^{w_1} \bar{l}_c(t)\,dt = \int_c^{w_1} e^{-\sum_{z=c}^{t} \mu_{z,z-c+s}}\,dt = \int_c^{w_1} e^{-\sum_{z=c}^{t} RR(PM)\mu_{z,z-c+s}}\,dt \tag{7}$$

where $\bar{l}_c$ is the survival function with the modified mortality rates and RR a function of (the change in) PM concentrations following Equation (4):

$$RR(PM) = (\beta PM) + 1$$

The absolute change in life expectancy per person is

$$\begin{aligned}
\Delta e_c &= \bar{e}_c - e_c \\
&= \int_c^{w_1} \bar{l}_c(t)\,dt - \int_c^{w_1} l_c(t)\,dt \\
&= \int_c^{w_1} e^{-\sum_{z=c}^{t} (\beta PM + 1)\mu_{z,z-c+s}}\,dt - \int_c^{w_1} e^{-\sum_{z=c}^{t} \mu_{z,z-c+s}}\,dt \\
&= \int_c^{w_1} (e^{-\sum_{z=c}^{t} \mu_{z,z-c+s}} e^{-\sum_{z=c}^{t} \beta PM \mu_{z,z-c+s}})\,dt - \int_c^{w_1} e^{-\sum_{z=c}^{t} \mu_{z,z-c+s}}\,dt \\
&= \int_c^{w_1} (e^{-\sum_{z=c}^{t} \mu_{z,z-c+s}} [e^{-\sum_{z=c}^{t} \beta PM \mu_{z,z-c+s}} - 1])\,dt \\
&= \int_c^{w_1} (l_c(t)[e^{-\sum_{z=c}^{t} \beta PM \mu_{z,z-c+s}} - 1])\,dt
\end{aligned} \tag{8}$$

This specification has the disadvantage that the RR function is part of the exponent of the e-function. In order to simplify, with

$$l_c(t) = e^{-\sum_{z=c}^{t} \mu_{z,z-c+s}},$$

the following substitution is permissible :

$$-\sum_{z=c}^{t} \mu_{z,z-c+s} = \ln l_c(t) \qquad (9)$$

Substituting (9) in (8) leads to

$$\Delta e_c = \int_c^{w_1} l_c(t)[e^{\beta * PM * \ln l_c(t)} - 1]dt \qquad (9')$$

To simplify further, the following linear approximation of (9') by means of a Taylor-approximation of degree 1 around 0 is used. The quality of the fit of this approximation is discussed below.

$$e^{(\beta * PM) \ln l_c(t)} - 1 \approx (\beta * PM) \ln l_c(t) \qquad (10)$$

Thus the absolute change in life expectancy per person of a cohort c in year s is

$$\Delta e_c = (\beta * PM) \int_c^{w_1} l_c(t) \ln l_c(t)dt = (\beta * PM) H_c \qquad (11)$$

where

$$H_c = \int_c^{w_1} l_c(t) \ln l_c(t)dt.$$

The change in life years for all persons of one cohort in grid cell x,y is obtained by multiplying Equation (11) by the size of the cohort $P_{c/x,y}$ and the length of the time interval for which demographic and mortality data are given. (For this study, data are available for five-year intervals.)

This leads to the change in life years lived for cohort c in grid cell x,y. As cohort data were obtained with reference to the aggregate national level, cohort size in a grid cell was calculated by weighting total population in a grid cell with the relative share of the given cohort in the national population:

$$\Delta L_c = P_{c/x,y} * \Delta e_t * i \qquad (12)$$

where

$$P_{c/x,y} = P_{c/national} * \frac{P_{total/x,y}}{P_{total/national}} \qquad (12')$$

where

ΔL_c	change in life years lived for cohort c in grid cell x,y
$P_{c/x,y}$	population in cohort c in grid cell x,y
$P_{c/national}$	national population in cohort c

$P_{total/x,y}$	total population in grid cell *x,y* (at least of age 30)
$P_{total/national}$	total national population (at least of age 30)
i	length of time interval

For all cohorts in a grid cell *x,y* the change in life years is expressed as the sum of the change in life years for the cohorts:

$$\Delta L_{x,y} = \sum_{c=w_0}^{w_1} \Delta L_c = i * (\beta * PM) * \frac{P_{total/x,y}}{P_{total/national}} \left(\sum_{c=w_0}^{w_1} H_c * P_{c/national} \right) \qquad (13)$$

where

w_0	first cohort considered (here 30)
w_1	last cohort considered (here 95)

Dividing (13) by total population at least of age 30 in grid cell *x,y* leads to the average change in life expectancy in grid cell *x,y*.

$$\Delta E_{x,y} = \frac{\sum_{c=w_0}^{w_1} \Delta L_c}{P_{total/x,y}} = i * (\beta * PM) \frac{\sum_{c=w_0}^{w_1} H_c * P_{c/national}}{P_{total/national}} \qquad (14)$$

In order to calculate the average change in life expectancy for a country *A*, the change in life years in all grid cells of a country divided by total population is computed:

$$\begin{aligned} \Delta E_A &= \frac{\sum_x \sum_y \Delta L_{xy}}{P_{total/nat.}} \\ &= \frac{i}{P_{total/nat.}} * \sum_x \sum_y [(\beta * PM_{x,y}) * \frac{P_{total/x,y}}{P_{total/nat.}} \sum_{c=w_0}^{w_1} (H_c * P_{c/nat.})] \\ &= \frac{i}{P^2_{total/nat.}} * \sum_x \sum_y [(\beta * PM_{x,y}) * P_{total/x,y} (\sum_{c=w_0}^{w_1} H_c * P_{c/nat.})] \end{aligned} \qquad (15)$$

where ΔE_A is the change in average life expectancy in country *A* expressed in years.

5.2.6 Error due to the linear approximation of the full model

As mentioned in the context of Equation 4, the methodology uses linear approximations for the hazard rate, i.e., of the relative risk and for calculating absolute changes in life expectancy according to

Equation 10. This greatly speeds up the calculations since the second term in Equation 15 containing H_c can be pre-calculated and does not need to be computed for each scenario and grid cell.

It turns out that the linear approximation to the full model described above is reasonably good for the estimation of impacts in Europe. Figure 5.1 shows the estimation error for the "Current legislation" scenario for all grid cells. No clear bias in either direction can be detected.

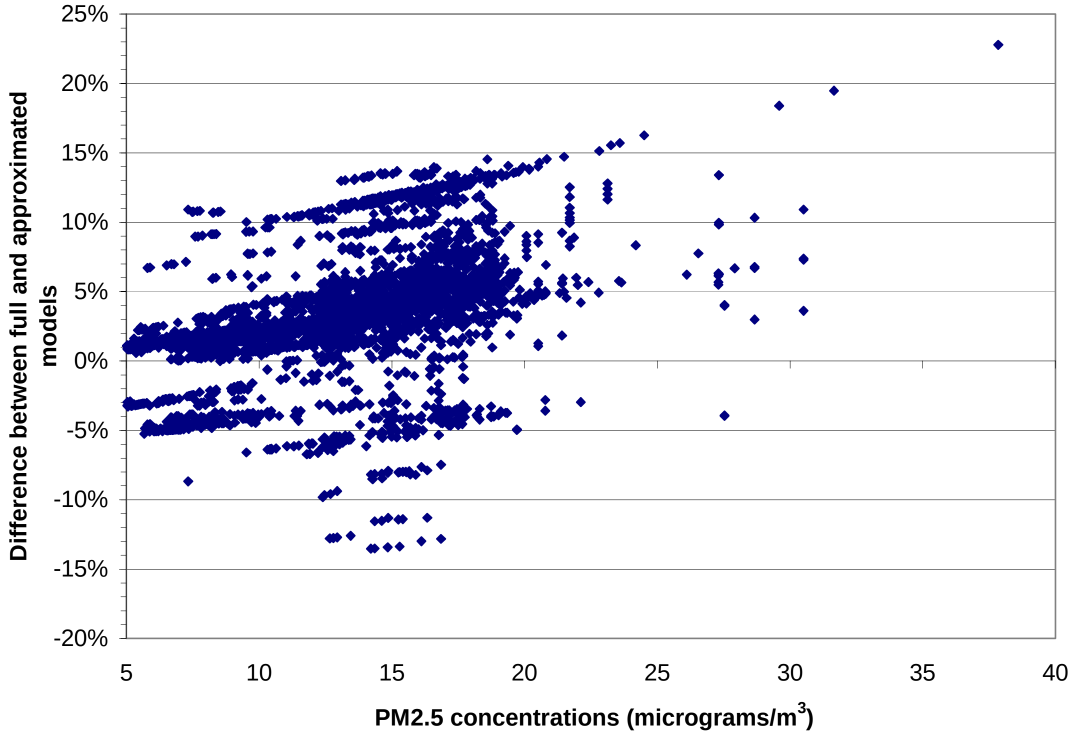

Figure 5.1: Approximation of the linear approximation model as a function of the change in PM for the CLE scenario in 2010

5.2.7 Transferability

A health impact assessment applies air pollution effect estimates derived from one (evidentiary) population to estimate impacts in another (target) population, based on the assumption that these estimates can be transferred. Care must be taken if one cannot assume that the contribution of various causes of death is similar, if the mixture of pollutants differs, if the baseline health statuses of the populations are not the same or if exposure ranges do not overlap.

Currently, only the cohort studies listed in Table 5.2 are available and provide the basis for numerous impact assessments. Since all but one of these cohort studies were conducted in the United States, the generalization of their results to populations in Europe and elsewhere is a concern. Recent studies have begun to explore effect modifiers that may explain the variation in air pollution effect estimates observed among locations in Europe and the United States (HEI, 2000; Katsouyanni *et al.*, 2001).

However, results for PM2.5 are not yet available and the present knowledge is quite limited, so that it is difficult to include other factors in a practical impact assessment at this point in time.

In its review of the RAINS methodology, the UN/ECE-WHO Task Force on Health (TFH, 2003) "*endorsed the decision to apply the relative risk for all causes of mortality estimated for the average exposure level in the extended American Cancer Society (ACS) cohort study as described by Pope et al. (2002). It was felt that this risk coefficient was a more appropriate choice than the estimates specific to the PM levels in the initial or final period of the follow-up in the ACS study, since there were indications that for some health end points, such as cardiopulmonary mortality, recent exposure was relevant, while for others, such as lung cancer, it could be assumed that exposure dating from both periods of exposure was important. Some participants noted that this choice was possibly biased towards underestimating the effects, since the population in the cohorts followed had an educational status above average in the United States, while the risk was higher for those with lower education. In addition, it was also noted that the estimate for relative risk from the ACS study was lower than from another available cohort study (the Six City Study). CIAM was invited to conduct sensitivity analysis using the relative risk based on the initial exposure level reported by Pope et al*.".

5.2.8 Extrapolations beyond the range of observational evidence

As pointed out by WHO (2001) caution must be used in extrapolating beyond the range of pollutant concentrations reported in the evidentiary study. The study of Pope *et al.* (2002) to which this assessment refers, covers annual mean PM2.5 concentrations from approximately 5[2] to 33.5 μg/m^3.

As will be explained below, the RAINS model relies only on dispersion calculations for the anthropogenic fraction of PM, but does not quantify the background contributions from natural sources (sea salt, wind blown dust, etc.). Thus, for the purpose of estimating the loss of life expectancy RAINS assumes that the relationship identified by Pope et al. (2002) holds for variations in $PM_{2.5}$ concentrations that are caused by changes in anthropogenic emissions. The effect of natural sources is neglected. Health effects are calculated only for the anthropogenic fraction without threshold in anthropogenic $PM_{2.5}$.

In its review of the RAINS methodology, the UN/ECE-WHO Task Force on Health (TFH, 2003) endorsed this approach and "recommended using only the anthropogenic contribution to PM2.5 mass; for this anthropogenic contribution, no no-effect level was assumed." *(The example calculations presented in Mechler et al., 2002 do not yet follow this rationale, but will be used as a sensitivity analysis to test the influence of the above mentioned assumptions).*

Similarly, there are 280 grid cells (5.2 percent of all total grid cells), where for 1990 PM2.5 concentrations are calculated to exceed the upper range of 33.5 μg/m^3 analyzed by Pope *et al.* (2002). For these situations the assumption is made that the linear response identified for the study domain does also hold, at least up to annual mean concentrations of 80 μg/m^3.

In its review of the RAINS methodology, the UN/ECE-WHO Task Force on Health (TFH, 2003) "*concluded that it was appropriate to extrapolate the concentration-response function linearly to higher concentrations than those of the evidentiary population. It was assumed that this choice would*

[2] The exact value of the lower range of PM2.5 concentrations was not published in Pope *et al.* (2002). In Figure 1 (Pope *et al.,* 2002: p. 1136) the lower value for PM2.5 concentrations for 1999-2000 is around 5 μg/m.[3] The exact value will be used once more data are made available from the Pope study.

not influence the results of the scenario analysis strongly, since it was expected that $PM_{2.5}$ concentrations at urban background locations would exceed the upper range of the ACS data only in a few cities in 2010 and onwards. IIASA was invited to conduct a sensitivity analysis using a log linear concentration-response relationship."

5.2.9 Assumptions for the preliminary implementation in the RAINS model

The preliminary implementation in the RAINS model employs the following assumptions:

- Mortality is calculated for exposure to $PM_{2.5}$.
- The rate of relative risk (RR) of 1.06 found by Pope *et al.* (2002) for the average period 1979-2000 for all-cause mortality.
- American RR applicable to Europe.
- The validity of the RR over the entire range of $PM_{2.5}$ that can be attributed to anthropogenic emissions.
- Only primary PM emissions and inorganic secondary PM is considered, which leads potentially to an underestimation of the 'real' effects.
- No effects from natural emissions.
- A linear extrapolation of risks beyond 35 $\mu g/m^3$ up to 80 $\mu g/m^3$
- No effects for younger than 30 years
- Risks are related to urban background concentrations of $PM_{2.5}$.
- Uniform urban background concentrations in each city.
- For each scenario constant exposure 2010-2080, cohorts followed up to end of their life time.

5.2.10 Uncertainties

Many of the data, models and assumptions used for the estimation of the impact have some uncertainty. This applies, *inter alia*,

- to the estimates of primary PM2.5 emissions in Europe,
- to the projections of future emission levels of PM and other pollutants in Europe,
- to calculations of the formation and atmospheric dispersion of primary and secondary aerosols in Europe,
- to estimates of ambient PM levels in urban air sheds,
- to the use of appropriate dose-response curves derived from epidemiological studies,
- to the question which property or chemical species of particulate matter is causally linked with mortality.

Each of these aspects is associated with considerable uncertainties. By linking this information, the methodology to estimate losses in life expectancy combines these uncertainties. Suutari *et al.* (2001) developed a methodology to propagate uncertainties through a similar chain of model calculations aiming at determining ecosystems protection from alternative emission control scenarios. It was shown that, as long as the uncertainties in different elements of the model chain (e.g., the estimates of emissions and of ecosystems sensitivities) are statistically independent from each other, uncertainties do not accumulate, but compensate each other to a large extent.

In principle, this methodology could equally well be applied to the calculation of losses in life expectancy to quantify uncertainties of the overall results, although in practice such an implementation would take considerable time and resources. At this stage a partial sensitivity analysis will be conducted using the upper and lower bounds of the 95 percent confidence interval of the relative risk function identified by Pope *et al.* (2002). Thus, the sensitivity analysis explores the losses in life expectancy resulting from relative risks of 1.02 and 1.11, compared to the central estimate of 1.06 per 10 µg/m3.

5.2.11 Other health effects of fine particulate matter

It is obvious that the approach - while appropriate for including the effects of PM on human health into the integrated assessment framework – does not yield an overall quantification of all effects related to exposure to particulate matter. Important effects which are currently not covered, but should eventually be taken into account in any cost benefit analysis, include infant mortality and morbidity outcomes.

While the modelling of infant mortality is at the present hampered by the lack of strong quantitative observational evidence, a number of concentration-response curves are available in the literature that would allow estimating morbidity impacts for various health endpoints. In practice, however, such calculations require sufficiently robust statistical information on base rates for the various health endpoints (e.g., asthma, hospital admissions, cough days, etc.) in an internationally consistent fashion. Advice from the health community strongly cautions against the use of the presently available health data for such an international assessment. While mortality statistics are undisputed, there are serious inconsistencies in the health statistics within Europe, which would discredit any calculations conducted on this basis. Thus, the RAINS model for the time being refrains from quantifying morbidity effects. In the context of CAFE, morbidity effects will be considered by the complementary cost-benefit analysis (http://europa.eu.int/comm/environment/air/cafe/activities/cba.htm). Along a parallel track, a research team at RIVM has started to develop an acceptable approach for quantifying morbidity impacts with the ultimate aim of introducing it into the RAINS model once it has achieved sensible results.

5.3 Modelling health-relevant source-receptor relationships for fine particles

5.3.1 Health-relevant metrics of air quality

The above approach quantifies changes in life expectancy as a function of changes in population exposure to fine particles. For their analysis the underlying epidemiological cohort studies employed annual mean concentrations of fine particles ($PM_{2.5}$) measured at fixed monitoring sites representative for urban background air sheds. To maintain formal consistency with the evidentiary studies from which relative risk rates are derived, an integrated assessment model also needs to connect mortality changes with the same air quality indicators. Thus, as long as the RAINS model relies on these evidentiary cohort studies, it needs to base its impacts estimates on annual mean concentrations of fine particles ($PM_{2.5}$) at background stations that are representative for the urban and rural population in Europe.

This consistency concern dictates the required output from atmospheric dispersion calculations in RAINS that provide the response of health-relevant air quality metrics towards changes in precursor emissions. Since epidemiological studies only provide *differences* in health impacts through comparisons of more polluted sites with cleaner locations, the emphasis of dispersion modelling needs to be more on reflecting these differences (e.g., due to differences in anthropogenic emissions) than in reproducing absolute PM levels.

5.3.2 The EMEP Eulerian model

Traditionally, RAINS calculations of the atmospheric dispersion of pollutants have been based on the EMEP model as a full-fledged atmospheric model. However, only recently, the new EMEP Eulerian model has become available, which allows for the first time to consider changes in fine particulate matter concentrations resulting from changes in anthropogenic emissions.

The peer review of the EMEP model identified a range of strengths and weaknesses that need to be considered when constructing source-receptor relationships for the integrated assessment of fine particulate matter. In particular, the review (TFMM, 2003) acknowledged that

- "*an excellent start had been made within EMEP to begin the challenging task of modelling the fate and behaviour of particulate matter across Europe by initially focusing on the mass fraction of particular components within PM_{10} and $PM_{2.5}$.*
- *There was high confidence in the model's ability to represent the broad spatial pattern of particulate sulphate across Europe, its trend and the role played by its long-range transport in providing the regional background levels required as an input to urban health impact studies.*
- *However, there are insufficient particulate nitrate and ammonium measurements available to provide an adequate test of the performance of the EMEP model.*
- *Confidence in the understanding of the mechanism of formation of secondary organic aerosols and of the quantification of some natural aerosol sources was so low that they had not been included in the EMEP model, leading to underestimations for PM_{10} and $PM_{2.5}$;*
- *Understanding was growing steadily in the quantification of primary particle emissions and this would lead eventually to increased confidence in the estimation of regional levels of primary emitted particles, thereby improving the assessment of PM_{10};*
- *Whilst there remained a significant fraction by mass unaccounted for between the model and observed PM_{10} and $PM_{2.5}$ levels, there was limited confidence in the model's ability to assess levels of PM_{10} and $PM_{2.5}$.*"

These findings have several implications for the use of this model within an integrated assessment framework:

- At the moment, the scientific peers do not consider the modelling of total particulate mass of the EMEP model (and of all other state-of-the-art models) as accurate and robust enough for policy analysis. Thus, one should not base an integrated assessment on estimates of total PM mass concentrations.
- The largest deficiencies have been identified in the quantification of the contribution of natural sources (e.g., mineral dust, organic carbon, etc.).
- The quantification of secondary organic aerosols (SOA) is not considered mature enough to base policy analysis on. A certain fraction of SOA is definitely caused by anthropogenic emissions, but some estimates suggest that the contribution from natural sources dominates total SOA. Clarification of this question is urgent to judge whether the inability of contemporary atmospheric chemistry models to quantify SOA is a serious deficiency for modelling the anthropogenic fraction of total PM mass.
- In contrast, the modelling of secondary inorganic aerosols is considered reliable within the usual uncertainty ranges. This applies especially to sulphur aerosols. The lack of formal validation of the nitrate calculations is explained by insufficient monitoring data with known accuracy; the model performs reasonably well for other nitrogen-related compounds.
- The validation of calculations for primary particles is hampered by insufficient observational data. Primary particles comprise a variety of chemical species, some of which (e.g., organic aerosols) originate from secondary particle formation too. Work at EMEP is underway to use improved emission inventories of black carbon, which are themselves only in a research phase, to use black carbon monitoring data as a tracer for emissions of primary particles. In principle, however, modelling of the dispersion of non-reactive substances like primary particles is generally considered as a not too ambitious undertaking. Thus, with some further evidence from EMEP/MSC-W on the performance of the Eulerian model for black carbon, an integrated assessment could rely on EMEP's dispersion calculations for primary particles over Europe.
- Thus, there are arguments that the present modelling capabilities allow quantification of the dispersion of (most of) the fine particles of anthropogenic origin. This would permit calculating changes in PM concentrations over Europe due to changes in anthropogenic emissions, and to estimate the health impacts that can be attributed to anthropogenic emission controls. On the other hand, it would not be possible to make any statements on the absolute level of PM mass concentrations, and subsequently not on the absolute health impacts of the total particle burden in the atmosphere. This limitation, however, does not seem to impose unbalanced restrictions on the overall analysis, since also the evidence from the available epidemiological studies does not allow drawing conclusions about the total health impacts. The RAINS approach as outlined in Section 5.2 acknowledges this by focusing on the differences in health effects in comparison with a baseline (reference) situation.

5.3.3 Source-receptor relationships for fine particulate matter

To quantify the health benefits of emission reductions, the RAINS integrated assessment model requires source-receptor relationships that describe changes in ambient levels of fine particles due to changes in the various anthropogenic precursor emissions from the various sources. The available concentration-response curves from the epidemiological studies relate differences in annual mean concentration of total $PM_{2.5}$ mass with observed changes in mortality. With the focus on changes in anthropogenic emissions, source-receptor relationships should describe the changed contributions of primary particles, secondary inorganic particles and secondary organic particles to total $PM_{2.5}$ mass due to changes in the emissions of SO_2, NO_x, VOC, NH_3 and primary PM.

With the caveats discussed above, the full-scale EMEP Eulerian atmospheric dispersion model is able to calculate these changes through respective "model experiments" (model runs with changed emissions). However, the computational complexity of the full EMEP model makes it impractical to operate the full EMEP model within the RAINS integrated assessment model, and makes it impossible to implement the optimisation approach, for which the "backwards" dispersion calculation is required. Thus, an attempt is made to identify appropriate "reduced form" source-receptor relationships that mimic the response of the full EMEP model in a computationally simple enough representation.

In their simplest form, reduced-form source-receptor relationships are linear, which allow simple matrix operations to compute air quality impacts from a given set of emission reductions or, conversely, to identify cost-effective control strategies that meet a set of air quality targets. In reality, physical and chemical processes are often rather complex and can in most cases only be fully represented by more complex, often non-linear, formulations. For the optimisation in an integrated assessment it is therefore of prime interest to understand to what extent such complex processes could be described by linearisations, which errors would be introduced by such linearisations and, if a linear description proves inadequate, which non-linear mathematical formulation could be developed that still allows efficient computation with the integrated assessment model.

For this purpose, a number of model experiments with the new EMEP Eulerian model have been designed that explore the response of computed $PM_{2.5}$ concentrations towards changes in the various precursor emissions. An initial round of model experiments with 87 model runs has been completed in January 2004, which now provides a first basis for the analysis of potential source-receptor relationships (Table 5.3). While it is interesting in itself to explore potential non-linearities in air quality responses to emission changes and to relate them to specific chemical processes, the analysis for the integrated assessment is driven by the need to make the mathematical model formulation as simple as possible, but not too simple. Thus, the analysis was limited to a well-defined range of emissions, corresponding to the practical scope of the policy analysis on further emission controls after the year 2010. Thus, the expected emissions from the baseline development in 2010, in which current air quality legislation will be implemented (the Current Legislation CLE scenario), mark the upper limit for the emission range. It is not envisaged that with the current legislation (e.g., the Emission Ceilings Directive) emissions would increase beyond these ceilings. The lower end is in principle determined by the "Maximum Technically Feasible Reduction" (MFR) scenario, in which all available control measures are introduced into the market following the natural renewal of the emission sources. To widen the scope of the analysis, an "Ultimately Feasible Reduction" scenario was considered, which ignores this penetration constraint and assumes full application of emission control measures at all sources. IIASA provided the quantified emission estimates for these scenarios, and MSC-W has produced the experiments with the EMEP Eulerian model.

The analysis explores the response towards emission changes of the various precursor emissions individually and collectively, for all of Europe and for individual countries, at different levels of pan-European emissions. Due to their emission densities, Germany, the Netherlands, UK and Italy have been selected as focal areas for detecting potential non-linearities.

Table 5.3: Overview of EMEP model experiments to explore non-linearities in source-receptor relationships. The following acronyms are used for the emission fields: CLE: Current legislation 2010, MFR: Maximum feasible reductions in 2010, UFR: Ultimately feasible reductions in 2020

	Changes in emissions						
	SOx	NOx + PPM	NH3	VOC	**NOX + SOX + PPM**	**VOC + NH3**	Nox + Sox + NH3+ VOC + CO + PPM
2010 CLE							R1 CLE_all
2010 MFR							R2 MFR_al
2010 UFR							R3 UFR_all
Italy as MFR Rest as CLE	R4 CLE_IT_MFR_NO	R5 CLE_IT_MFR_SP	R6 CLE_IT_MFR_NH	R7 CLE_IT_MFR_VOC	R8 CLE_IT_MFR_SNP	R9 CLE_IT_MFR_NVO	R10 CLE_IT_MFR_ALL
Italy as UFR Rest as CLE	R11 CLE_IT_UFR_NO	R12 CLE_IT_UFR_SP	R13 CLE_IT_UFR_NH	R14 CLE_IT_UFR_VOC	R15 CLE_IT_UFR_SNP	R16 CLE_IT_UFR_NVO	R17 CLE_IT_UFR_ALL
Italy as CLE Rest as UFR	R18 UFR_IT_CLE_NO	R19 UFR_IT_CLE_SP	R20 UFR_IT_CLE_NH	R21 UFR_IT_CLE_VOC	R22 UFR_IT_CLE_SNP	R23 UFR_IT_CLE_NVO	R24 UFR_IT_CLE_ALL

Netherlands as MFR Rest as CLE	R25 CLE_NL_ MFR_NO	R26 CLE_NL_ MFR_SP	R27 CLE_NL_MFR_NH	R28 CLE_NL_ MFR_VOC	R29 CLE_NL_ MFR_SNP	R30 CLE_NL_ MFR_NVO	R31 CLE_NL_ MFR_ALL
Netherlands as UFR Rest as CLE	R32 CLE_NL_ UFR_NO	R33 CLE_NL_ UFR_SP	R34 CLE_NL_UFR_NH	R35 CLE_NL_ UFR_VOC	R36 CLE_NL_ UFR_SNP	R37 CLE_NL_ UFR_NVO	R38 CLE_NL_ UFR_ALL
Netherlands as CLE Rest as UFR	R39 UFR_NL_ CLE_NO	R40 UFR_NL_ CLE_SP	R41 UFR_NL_CLE_NH	R42 UFR_NL_CLE_VOC	R43 UFR_NL_ CLE_SNP	R44 UFR_NL_ CLE_NVO	R45 UFR_NL_ CLE_ALL
United Kingdom as UFR Rest as CLE	R46 CLE_GB_ MFR_NO	R47 CLE_GB_ MFR_SP	R48 CLE_GB_MFR_NH	R49 CLE_GB_MFR_VOC	R50 CLE_GB_ MFR_SNP	R51 CLE_GB_ MFR_NVO	R52 CLE_GB_ MFR_ALL
United Kingdom as UFR Rest as CLE	R53 CLE_GB_ UFR_NO	R54 CLE_GB_ UFR_SP	R55 CLE_GB_UFR_NH	R56 CLE_GB_UFR_VOC	R57 CLE_GB_ UFR_SNP	R58 CLE_GB_ UFR_NVO	R59 CLE_GB_ UFR_ALL
United Kingdom as CLE Rest as UFR	R60 UFR_GB_ CLE_NO	R61 UFR_GB_ CLE_SP	R62 UFR_GB_CLE_NH	R63 UFR_GB_CLE_VOC	R64 UFR_GB_ CLE_SNP	R65 UFR_GB_ CLE_NVO	R66 UFR_GB_ CLE_ALL
Germany as MFR Rest as CLE	R67 CLE_DE_ MFR_NO	R68 CLE_DE_ MFR_SP	R69 CLE_DE_MFR_NH	R70 CLE_DE_MFR_VOC	R71 CLE_DE_ MFR_SNP	R72 CLE_DE_ MFR_NVO	R73 CLE_DE_ MFR_ALL
Germany	R74	R75	R76	R77	R78	R79	R80

as UFR Rest as CLE	CLE_DE_ UFR_NO	CLE_DE_ UFR_SP	CLE_DE_UFR_NH	CLE_DE_UFR_VOC	CLE_DE_ UFR_SNP	CLE_DE_ UFR_NVO	CLE_DE_ UFR_ALL
Germany as CLE Rest as UFR	R81 UFR_DE_ CLE_NO	R82 UFR_DE_ CLE_SP	R83 UFR_DE_CLE_NH	R84 UFR_DE_CLE_VOC	R85 UFR_DE_ CLE_SNP	R86 UFR_DE_ CLE_NVO	R87 UFR_DE_ CLE_ALL

Due to the short time since model results became available, only initial findings can be presented here.

The following graphs compare changes of calculated aerosol concentrations for the German grid cells (red crosses) and other European receptors (black crosses) resulting from changes in German emissions.

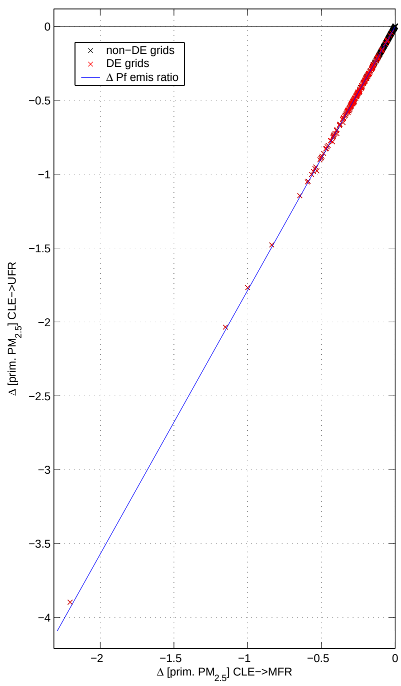

Figure 5.2: Changes in ambient PM2.5 concentrations from a CLE-MFR reduction of German $PM_{2.5}$ primary PM emissions versus changes in ambient PM2.5 concentrations from a CLE-UFR reduction of primary $PM_{2.5}$ emissions in Germany

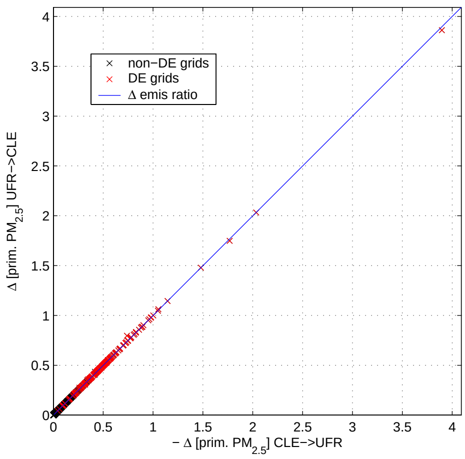

Figure 5.3: Differences in annual mean concentrations of ambient PM2.5 resulting from a change of German primary PM2.5 emissions from UFR to CLE with all other emissions at UFR versus a change of German primary PM2.5 emissions from CLE to UFR versus with all other emissions at CLE.

Figure 5.2 plots the changes in annual mean $PM_{2.5}$ mass concentrations due to reducing German primary $PM_{2.5}$ emissions from CLE to MFR (x-axis) versus the changes resulting from emission reductions from CLE to UFR. The line indicates the ratio of emission changes, i.e., (CLE-UFR)/(CLE-MFR). For primary particles all changes in PM concentrations are entirely proportional to changes in emissions, i.e., there is a linear response of annual mean $PM_{2.5}$ concentrations towards changes in primary $PM_{2.5}$ emissions modelled by the EMEP Eulerian model. As shown in Figure 5.3, this linearity is independent from the level of pollutants, i.e., the same response occurs if changes are implemented from the CLE levels or around the UFR level.

The situation is more complex for secondary inorganic aerosols. As demonstrated in Figure 5.4, there is a linear relation between changes in SO_2 emissions and changes in ambient concentrations of secondary inorganic aerosols. The response is slightly dependent on the overall level of pollution (Figure 5.5), but this bilinearity does not seem dramatic.

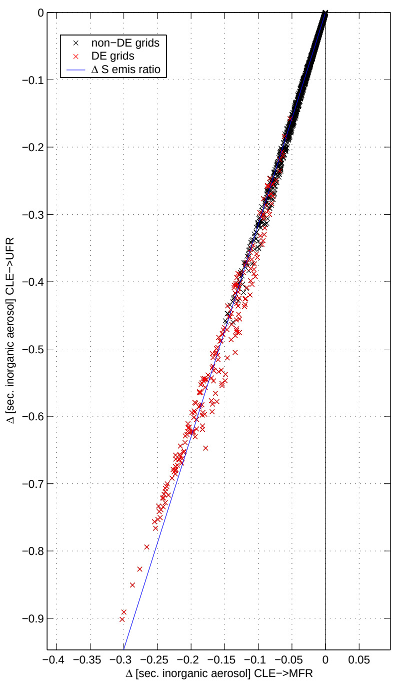

Figure 5.4: Changes in ambient secondary inorganic aerosol concentrations from a CLE-MFR reduction in SO_2 emissions versus changes in secondary inorganic aerosol concentrations from a CLE-UFR reduction of SO_2 emissions in Germany

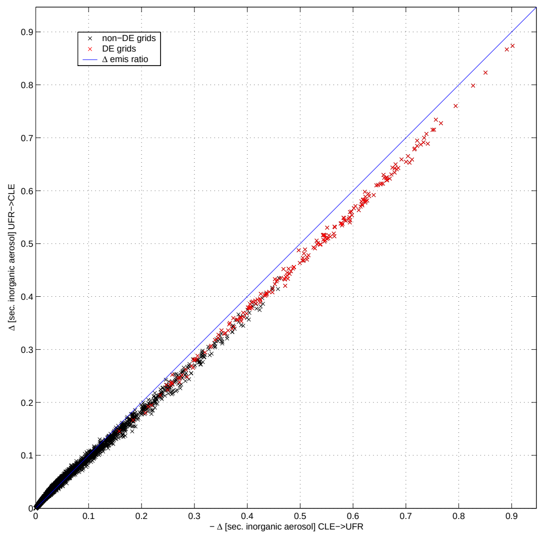

Figure 5.5: Differences in annual mean concentrations of ambient secondary inorganic aerosols resulting from a change of German SO_2 emissions from UFR to CLE with all other emissions at UFR versus a change of German SO_2 emissions from CLE to UFR with all other emissions at CLE.

Changes in NO_x emissions, however, result in non-linear responses in secondary inorganic aerosols (Figure 5.6). A unit of NO_x reduced in the range from CLE to MFR causes less decline in secondary inorganic aerosols than a unit reduced over the range from CLE to UFR. The deviation from linearity is larger for receptor sites close to the sources (the red crosses indicating the German sites) than at more distant receptors. The fact that the differences can be approximated by a linear function suggests applicability of a quadratic function to describe this response. As shown in Figure 5.7, the non-linearity also depends on the absolute level of emissions: the same delta NO_x emissions reduced at CLE levels (of all pollutants) causes larger changes in secondary inorganic aerosol concentrations than at the UFR level. It should be noted that this preliminary analysis is conducted for total inorganic aerosols; some of the compounds, especially nitrates, fall however into the coarse fraction larger than 2.5 μm, so that the analysis should be repeated for secondary inorganic aerosols in the fine fraction only.

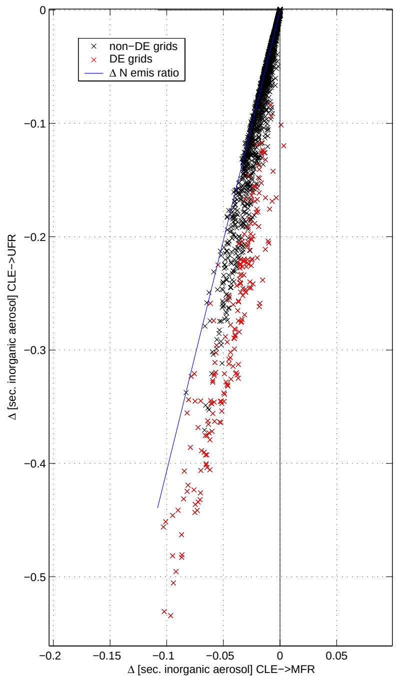

Figure 5.6: Changes in ambient PM2.5 concentrations from a CLE-MFR emission reduction versus changes in ambient PM2.5 concentrations from a CLE-UFR emission reduction in Germany

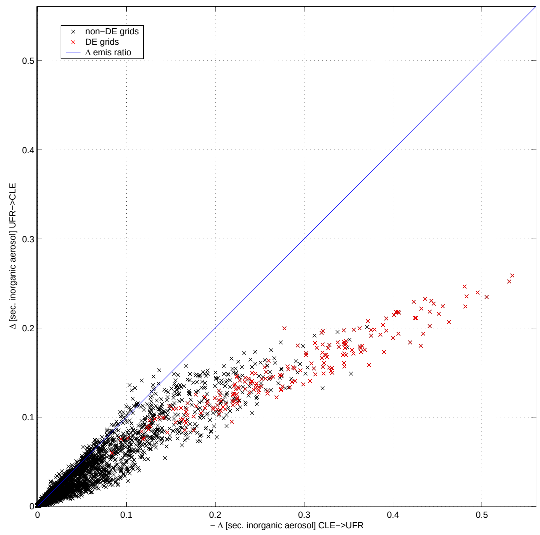

Figure 5.7: Differences in annual mean concentrations of ambient secondary inorganic aerosols resulting from a change of German NO_x emissions from UFR to CLE with all other emissions at UFR versus a change of German NO_x emissions from CLE to UFR with all other emissions at CLE.

A similar non-linear response as for NO_x can be diagnosed for changes in NH_3 emissions (Figure 5.8, Figure 5.9). Again, the non-linearities depend on the general pollution level and the analysis needs to be repeated for secondary inorganic aerosols in the fine fraction.

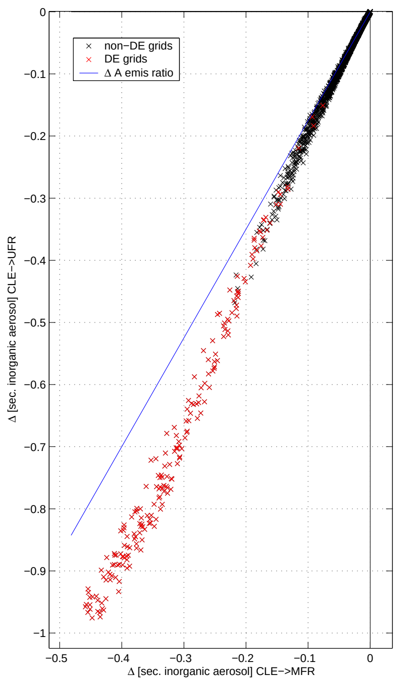

Figure 5.8: Changes in ambient secondary inorganic aerosol concentrations from a CLE-MFR reduction of ammonia emissions versus changes in ambient secondary inorganic aerosol concentrations from a CLE-UFR ammonia emission reduction in Germany

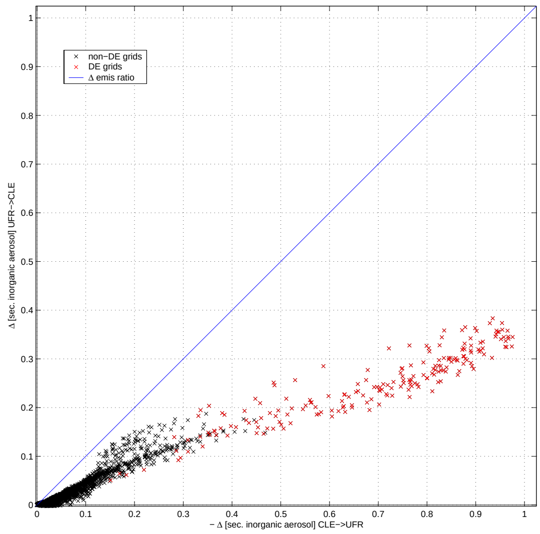

Figure 5.9: Differences in annual mean concentrations of ambient secondary inorganic aerosols resulting from a change of German NH_3 emissions from UFR to CLE with all other emissions at UFR versus a change of German NH_3 emissions from CLE to UFR with all other emissions at CLE.

The EMEP Eulerian model also shows a response of secondary inorganic aerosols (SIA) towards changes in VOC emissions although VOC itself is not a direct precursor of SIA. The response is however linear and relatively small (Figure 5.10), but depends on the general level of pollution (Figure 5.11). Possible explanations relate to modifications in the oxidizing capacity of the atmosphere, which would lead to faster oxidation of sulphates and nitrates.

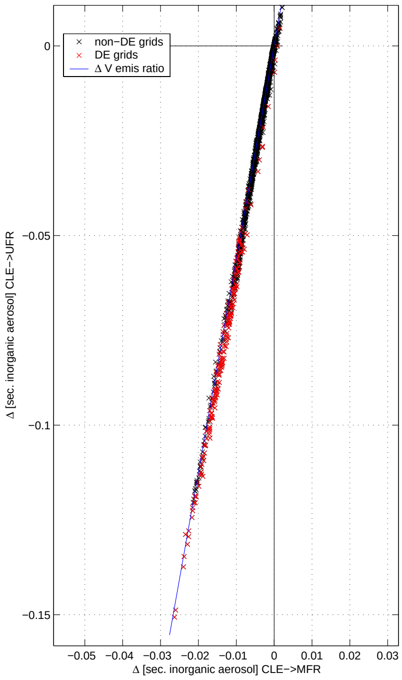

Figure 5.10: Changes in ambient SIA concentrations from a CLE-MFR reduction of VOC emissions versus changes in ambient SIA concentrations from a CLE-UFR VOC emission reduction in Germany

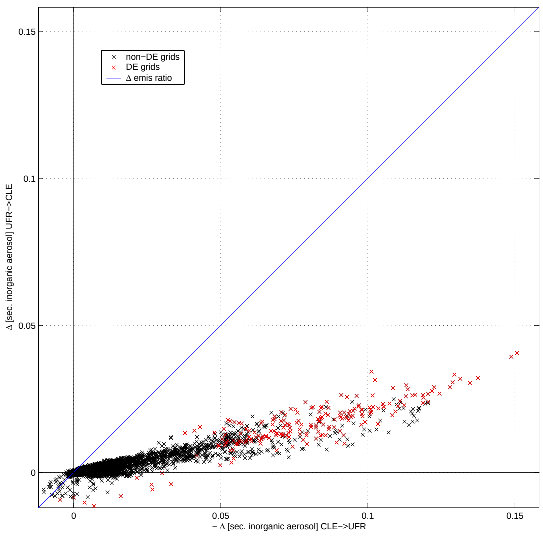

Figure 5.11: Differences in annual mean concentrations of ambient secondary inorganic aerosols resulting from a change of German VOC emissions from UFR to CLE with all other emissions at UFR versus a change of German VOC emissions from CLE to UFR with all other emissions at CLE

The above analysis identifies for NO_x, NH_3 and VOC clear non-linearities, if these emissions are reduced in isolation. In practice, however, realistic emission control scenarios will always have simultaneous effects on multiple pollutants. However, deviations from linearity are minimal, if all emissions are reduced simultaneously, and this linearity holds over the full range of analysed emissions, at least for more distant receptors (Figure 5.12). It should be noted that, while all examples presented above refer to a change in German emissions, the same pictures emerge for the other countries (Netherlands, the UK and Italy) that have been analysed up to now.

As explained above, the emphasis of the RAINS analysis for health impacts will be on changes in total $PM_{2.5}$ mass concentrations. Thus, non-linearities identified for secondary inorganic aerosols should be seen in the context of total $PM_{2.5}$ concentrations. From this perspective, deviations from linearity are rather small and almost independent of the pollution level (Figure 5.13).

It remains to be determined how relevant the detected non-linear model responses will be for the practical policy assessment (i.e., for the magnitude of emission changes that will be taken into closer consideration) and what bias would be introduced into the calculation by a simplified representation of this phenomenon. Finally, a practical mathematical formulation will need to be developed so that the assessment of health impacts in the RAINS model can be conducted with sufficient accuracy.

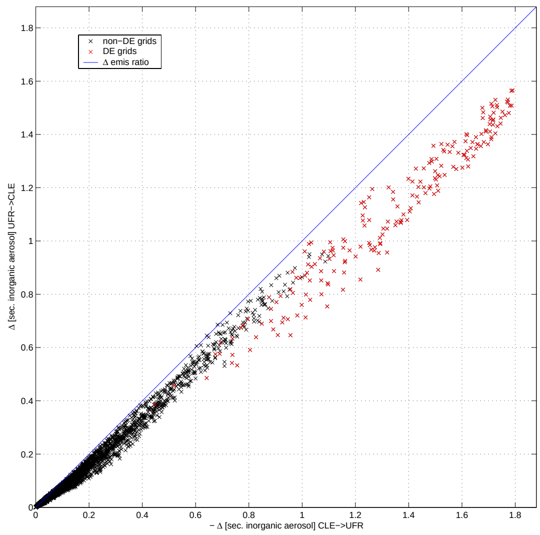

Figure 5.12: Differences in annual mean concentrations of ambient secondary inorganic aerosols resulting from a change of German SO_2, NO_x, VOC, NH_3 and primary PM2.5 emissions from UFR to CLE with all other European emissions at UFR versus a change of the German emissions from CLE to UFR with all other European emissions at CLE

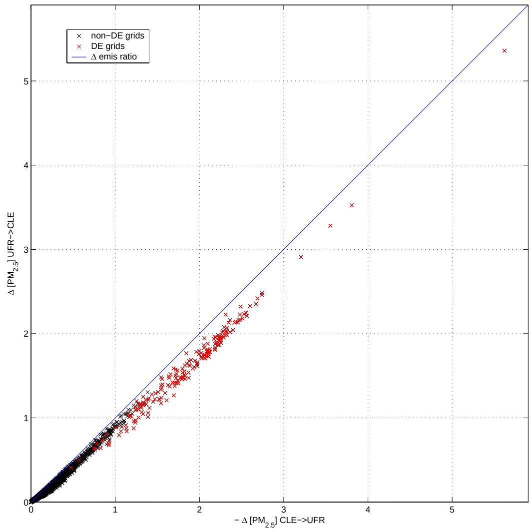

Figure 5.13: Differences in annual mean concentrations of ambient $PM_{2.5}$ resulting from a change of German SO_2, NO_x, VOC, NH_3 and primary PM2.5 emissions from UFR to CLE with all other European emissions at UFR versus a change of the German emissions from CLE to UFR with all other European emissions at CLE

5.3.4 Assessment of urban air quality

The underlying epidemiological studies detected and quantified relationships between mortality and characteristic background concentrations of fine particulate matter. Thus, in order to accurately apply their findings for a health impact assessment, it will be necessary to target the characteristic exposure for the urban and rural population in Europe. The EMEP Eulerian model provides estimates of regional air quality with a 50*50 km resolution, which can be considered as representative for rural air sheds. However, the majority of the European population lives in urban areas, and the health impact assessment needs to reflect the higher concentrations of particles in urban air.

5.3.4.1 The City-Delta project

To acquire deeper insight into the factors that deteriorate air quality in cities, IIASA together with the Institute for Environment and Sustainability of the Joint Research Centre (Ispra), MET.NO, EUROTRAC-2 and CONCAWE, has initiated the City-Delta model intercomparison (http://rea.ei.jrc.it/netshare/thunis/citydelta/). The aim of this exercise is to conduct a systematic comparison between regional scale and local scale dispersion models to identify and quantify the factors that lead to systematic differences between air pollution in urban background air and rural background concentrations.

City-Delta explores

- systematic differences (deltas) between rural and urban background AQ,
- how these deltas depend on urban emissions and other factors,
- how these deltas vary across cities, and
- how these deltas vary across models.

Based on the findings of City-Delta, functional relationships will be developed that allow the estimation of urban levels of pollution (PM2.5) as a function of rural background concentrations and local factors. The City-Delta analysis addresses the response of health-relevant metrics of pollution exposure (i.e., long-term concentrations with or without thresholds) towards changes in local and regional precursor emissions, including the formation of secondary aerosols. This enables the generic analysis of urban air quality for a large number of European cities based on information available in the RAINS model framework.

City-Delta provides harmonized emission inventories, meteorological conditions and observational data and explores the changes in air quality for seven emission control scenarios in eight European cities. It is important that the participating models apply different spatial resolutions, i.e., some of the models operate with a 50 km resolution and are thus directly comparable to the EMEP model (which is also participating), while others use finer resolution in the urban model domains. Thus, it is now possible to directly compare results from European scale models with finer resolved urban dispersion models and to search for systematic differences.

In its first phase, 20 modelling teams participated in City-Delta and produced for PM in total 143 model runs, each of them covering the full 12-months period of 1999 (Table 5.4). The focus of the City-Delta project on health-relevant air quality metrics gave a major impetus to many participating models that used to operate for selected short-term episodes only to extend their modelling capacities to full 12 months calculations. As a result, there is now a large ensemble of dispersion models available in Europe that can be used for health impact assessment according to the WHO recommended methodology.

The first phase of City-Delta clearly revealed serious deficiencies in the present scientific ability to accurately model observed PM mass concentrations. All chemistry models that simulate the fate of the various chemical components of PM result in serious underestimates of observed PM mass concentrations. However, models agree to a large extent on the fate of anthropogenic primary particles and secondary inorganic aerosols. It should be noted that also the lack of quality controlled monitoring data, especially for $PM_{2.5}$, puts a serious limit to the validation of the model results.

Table 5.4: Models participating in City-Delta (O .. ozone calculations, P .. particle calculations)

City	Berlin			Copenhagen			Katowice			London			Milan			Paris			Prague		
Resolution	5 km	10 km	50 km	5 km	10 km	50 km	5 km	10 km	50 km	5 km	10 km	50 km	5 km	10 km	50 km	5 km	10 km	50 km	5 km	10 km	50 km
THOR	OP	OP	OP	OP	OP	OP	OP	OP	OP	OP	OP	OP	??	??	??	??	??	??	??	??	??
MOCAGE	-	-	-	-	-	-	-	-	-	-	-	-	-	O	O	-	O	O	-	-	-
CALGRID	-	-	-	-	-	-	-	-	-	-	-	-	O	-	-	-	-	-	-	-	-
CAMx	-	-	-	-	-	-	-	-	-	-	-	-	O	-	-	-	-	-	-	-	-
REM3	OP	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
STEM	-	-	-	-	-	-	-	-	-	-	-	-	O	?	-	-	-	-	-	-	-
SMOG	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	O	O	-
MCCM	OP	-	OP	-	-	-	OP	-	OP	-	-	-	-	-	-	-	-	-	-	-	-
CHIMERE	OP	-	OP	-	-	-	O	-	O	OP	-	OP	OP	-	OP	OP	-	OP	O	-	O
OFIS		OP			-			-			OP			OP			OP			OP	
MUSE	-	O	-	-	-	-	-	-	-	-	O	-	-	O?	-	-	O	-	-	O	-
AURORA	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
UAM-IV	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
EPISODE	-	O	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
EMEP	-	-	OP	-	-	OP	-	-	OP	-	-	OP	-	-	OP	-	-	OP	-	-	OP
LOTOS	O	O	O	-	-	-	-	-	-	-	-	-	-	-	-	OP	OP	OP	-	-	-
EUROS		O																			
TRANSCHIM	O																				
MUSCAT	OP																				
POLSKA	-	-	OP	-	-	OP	-	-	OP	-	-	OP	-	-	OP	-	-	OP	-	-	OP

While models do not perform well in reproducing the observed total PM mass, the limited available monitoring data strengthen their credibility for modelling anthropogenic fractions of PM, at least from primary emissions and from secondary inorganic aerosols. Based on this assertion, City-Delta 1 compared the responses of regional and urban scale dispersion models towards changes in regional precursor emissions (Figure 5.14). Models also agree that for urban areas there seems to be in the year

2010 less practical scope for further reductions of PM than the expected improvements between 1999 and 2010.

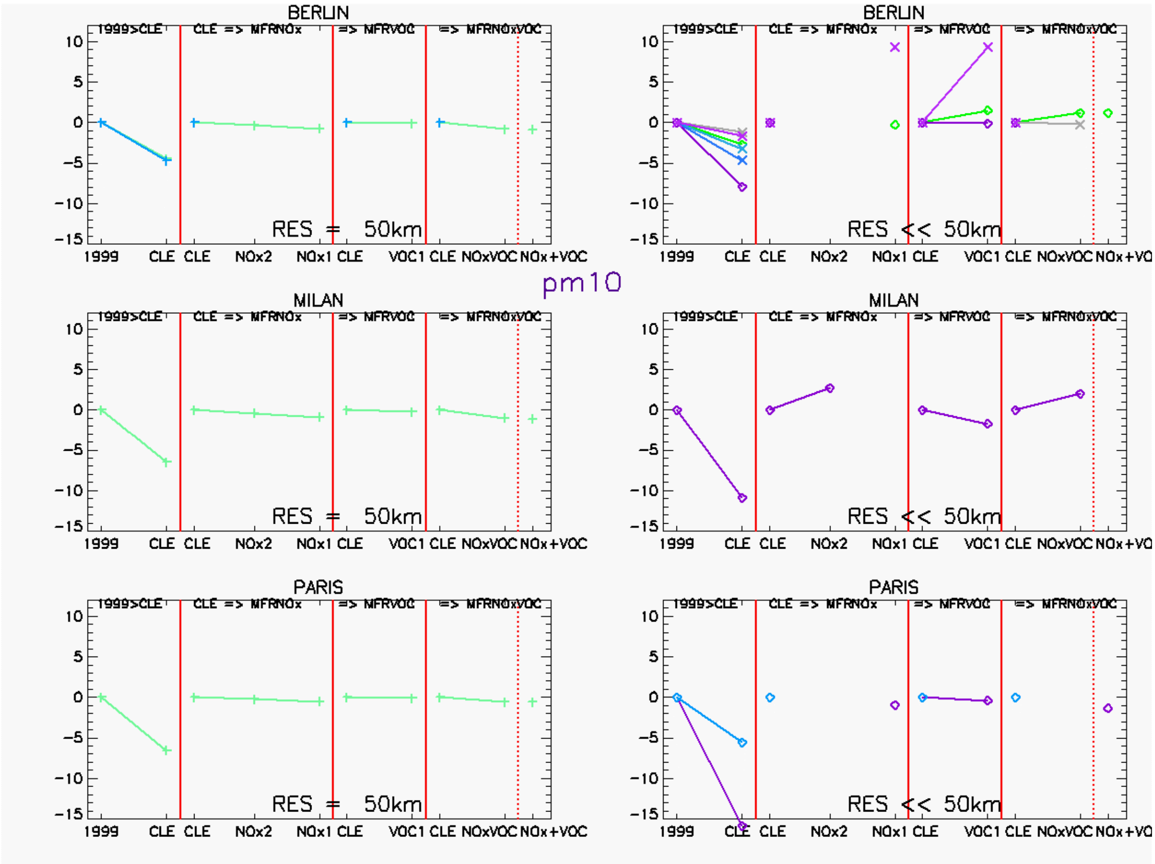

Figure 5.14: Summary of City-Delta Phase 1 model responses of urban PM_{10} levels (annual mean concentrations) towards changes in the various precursor emissions for Berlin, Milan and Paris. The left panel shows responses of regional scale models (50 km resolution), while the right panel shows responses of fine scale models. Four “deltas” are presented: (1) the change between emissions of 1999 and CLE 2010; (2) the change from CLE 2010 to MFR NO_x; (3) the change from CLE 2010 emissions to MFR for VOC, and (4) the change from CLE 2010 to MFR for NO_x and VOC. Different models are displayed in different colours.

As a general finding it can be shown that PM concentrations modelled for cities are systematically higher than those in rural background air. The increase depends on the size fraction considered (there is a larger difference for PM_{10} than for $PM_{2.5}$) and on the emission densities within the cities. All models participating in Phase 1 of City-Delta agree that a large fraction of fine particles present in urban background air (especially $PM_{2.5}$) originate from outside the cities, and in many cases from long-range transport sources several hundred kilometres away. Thus, the boundary conditions of urban model calculation dominate the results also within the cities.

This finding, which is also supported by monitoring data (Putaud *et al.*, 2002), is important for the integrated assessment of European emission control strategies, because it underlines the relevance of Europe-wide emission control efforts also for the improvement of air quality in the cities.

Phase 1 of City-Delta has identified a number of factors that lead to differences in model results. Some of them are related to different model concepts and are therefore interesting from a scientific standpoint, but also a number of mistakes in input data (especially emission inventories), data handling or model routines have been identified. Many of the originally large discrepancies could be reduced by thorough quality control. Despite these improvements, the numerical results from City-Delta 1 for particulate matter were not considered sufficiently robust to serve as a basis for serious policy analysis. Thus, a second phase of City –Delta has been launched, which focuses specifically on particulate matter. This phase is expected to be completed in spring 2004 and will provide input to the RAINS modelling of urban air.

5.3.5 Modelling urban particulate matter in RAINS

While the findings of City-Delta 2 are not yet available, a concept has been developed on how to assess with the RAINS model levels of particulate matter in European urban areas. Obviously, this work is in progress, and might change if counter-evidence emerges in the coming months.

Also for urban areas, RAINS will use annual mean concentrations of $PM_{2.5}$ at urban background stations to quantify impacts on life expectancy. With this focus, the concept is based on the analysis of available observational data – supported by the findings of City-Delta 1 – that a large fraction of $PM_{2.5}$ in urban background air originates from long-range transport. Conceptually, this fraction will be calculated by the European scale Eulerian EMEP model for the 50*50 km grid cell in which a city is located. RAINS will then assess the extra contribution from urban sources. Preliminary analysis of monitoring data shows a clear relation between the magnitude of this urban signal and the emission density within the city. An example for the UK finds a rather consistent linear relation between observed annual mean PM_{10} levels at the various monitoring stations and the emission densities in the 5*5 km box around each station (Figure 5.15). Correlations are best if emission densities are averaged over 5*5 km rather than over 1*1, 3*3 or 10*10 km, and if only emissions from road transport are considered. As a sideline, it is interesting to note that such relations emerge also for roadside sites, though obviously with a steeper slope.

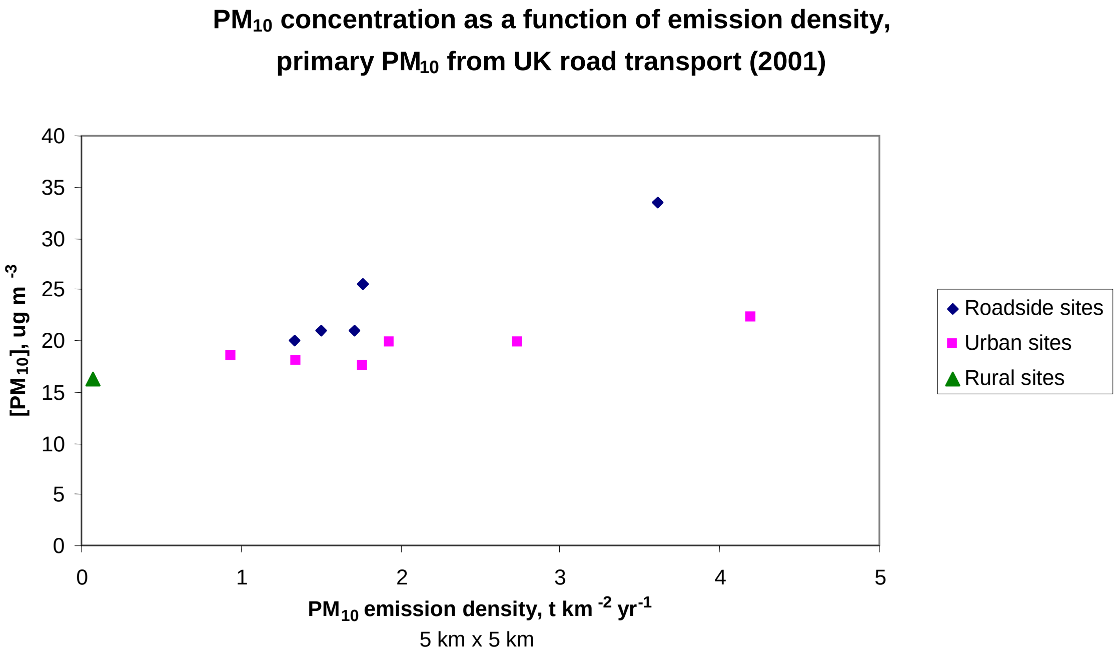

Figure 5.15: Relation between measured PM10 concentrations within London and the emission density of PM10 from road transport

The influence of traffic emissions on the urban signal of $PM_{2.5}$ is supported by monitoring results from the Austrian AUPHEP project (Puxbaum *et al.*, 2003). This project measured PM daily over 12 months at twin sites in and around Vienna and provides a chemical analysis of the various size fractions. As shown in Figure 5.16, the higher $PM_{2.5}$ concentrations in the city can mainly be found for black carbon, organic carbon and ammonium sulphate. While the ammonium sulphate increase in the city remains to be explained in the absence of major SO_2 sources in Vienna, the other components are a clear fingerprint of traffic-related emission sources. No significant differences between the urban and rural site were found for the other chemical species.

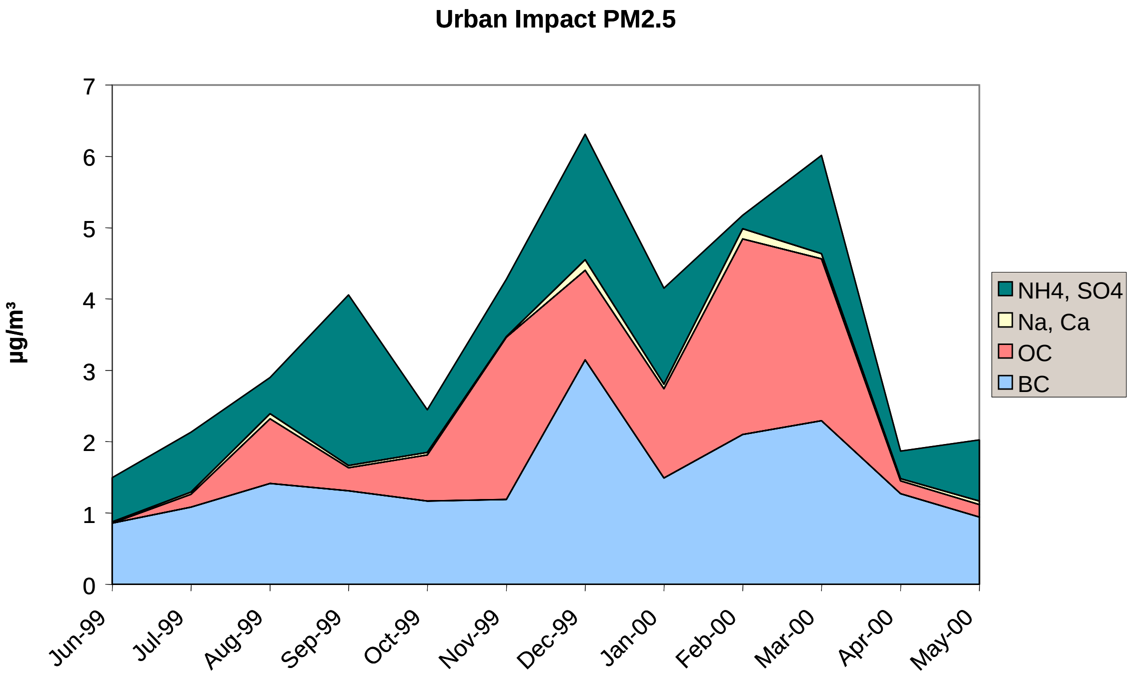

Figure 5.16: Urban impacts of PM2.5 in Vienna June 1999-May 2000, by chemical composition (i.e., difference between the $PM_{2.5}$ concentrations measured at the urban and the rural monitoring sites). Source: Puxbaum *et al.*, 2003

For comparison, Figure 5.17 shows the same analysis for the coarse fraction of PM, i.e., PM_{10}-$PM_{2.5}$. There is a strong signal of chemically undetermined species in the winter, presumably mineral dust from roads (gravel).

It is also interesting to note that $PM_{2.5}$ concentrations in Vienna have been greater than those at the pristine rural site (annual mean of 18 $\mu g/m^3$) by between 1.5 $\mu g/m^3$ in the summer and up to 6 $\mu g/m^3$ in the winter, with an annual average difference of about 4 $\mu g/m^3$. This confirms the importance of long-range transport also within the urban areas.

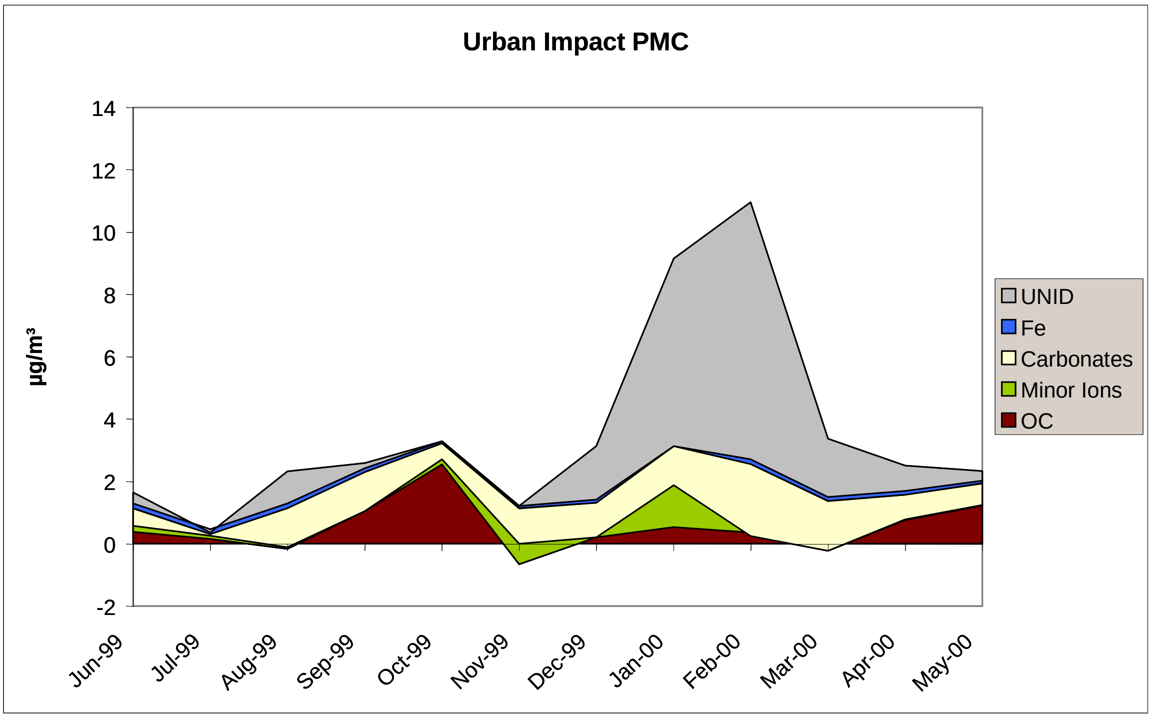

Figure 5.17: Urban impacts of the coarse PM (PM_{10}-$PM_{2.5}$) fraction in Vienna June 1999-May 2000, by chemical composition (i.e., difference between the PMc concentrations measured at the urban and the rural monitoring sites). Source: Puxbaum *et al.*, 2003

For the RAINS model it is planned to derive a relationship between emission densities within cities (possibly limited to transport sources) and the urban increment according to Figure 5.15. Work is underway to collect chemical and size-resolved monitoring data for additional cities. It is conceivable that additional factors need to be taken into account, such as the total size of a city or meteorological or topographic factors. City-Delta model results will help to test this hypothesis for various cities in Europe and derive a representative function.

With this function, regional scale PM concentrations as computed from the EMEP model can then be adjusted for urban areas, taking into account the emission densities, which are influenced, *inter alia*, by the amount of traffic emissions and thus by the level of emission controls applied to transport sources. While there are no consistent and reliable emission inventories available for the more than 200 cities in Europe, it is expected that the information available in RAINS, i.e., population data for each city, city areas, characteristic emission factors for transport sources, fleet composition, application of emission control measures, etc. should help to construct appropriate surrogates for this calculation.

5.4 Uncertainties

As explained above, many aspects load any estimate of health impacts of particles with significant uncertainties. For quantification of the health-relevant air quality changes resulting from emission changes, the general imperfections of dispersion modelling for fine particles cannot be eliminated in the near future, and additional uncertainties originate from lack of solid understanding of all emission sources.

While the specific approach for uncertainty treatment within the integrated assessment model can only be designed once the model approach has been ultimately decided (i.e., after all results from the EMEP dispersion model and City-Delta are finally available), preparatory actions have been taken to derive quantified estimates of the uncertainties of the various elements in the model chain. City-Delta by its design provides, *inter alia*, information about the extent of agreement and disagreement among the available state-of-the art urban dispersion models.

5.4.1 The Euro-Delta project

To derive similar information for regional scale source-receptor relationships, IIASA together with the Institute for Environment and Sustainability of the Joint Research Centre (Ispra), MET.NO, EUROTRAC-2 and CONCAWE, has initiated the Euro-Delta model intercomparison (http://rea.ei.jrc.it/netshare/thunis/eurodelta/). The aim of this exercise is to conduct a systematic comparison of regional scale dispersion models to judge the performance of state-of-the-art regional scale dispersion models in relation to health- and policy-relevant model output.

Five European scale dispersion models, including the EMEP Eulerian model, participate in this intercomparison (Table 5.5), which analyses model responses for PM and ozone for seven emission control scenarios.

Table 5.5: Participating models in Euro-Delta

Model	Contact person	Affiliation
LOTOS	P. Builtjes	TNO-MEP, (NL)
REM3/CALGRID	R. Stern	FUB, (D)
CHIMERE	C. Honore L. Rouil	INERIS, (F)
Unified EMEP	L. Tarrason	EMEP/MSC-W, (N)
MATCH	J. Langner	SMHI (S)
MODELS-3	I. Rodgers	INNOGY, (GB)

As of February 2004, most models have computed the emission scenarios, and the evaluation and interpretation of the results has started. As in City-Delta, the apparent lack of performance for reproducing observed PM mass concentrations is obvious for all models (Figure 5.18). There is more agreement among models for sulphates (Figure 5.19).

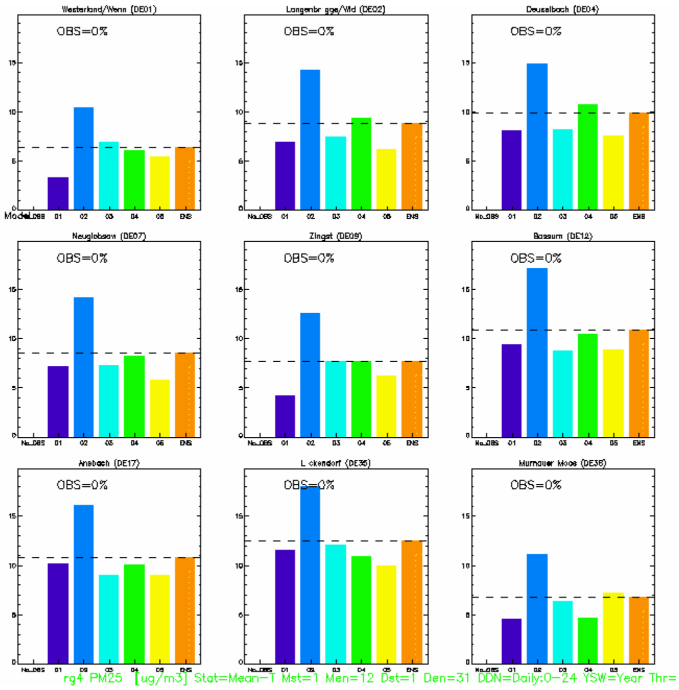

Figure 5.18: Annual mean concentrations of $PM_{2.5}$ (in µg/m^3) as computed by the Euro-Delta models for the German EMEP monitoring stations. No observational data is available for these stations. The EMEP model is Model04 printed in green.

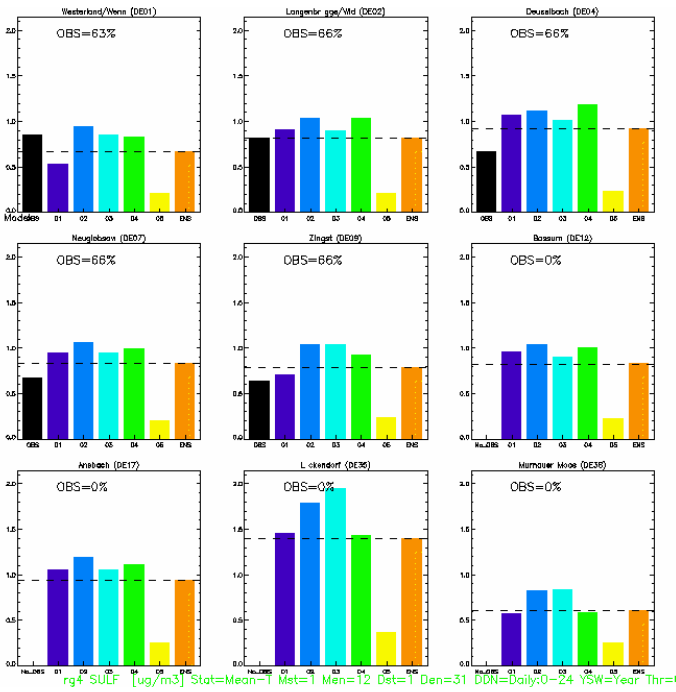

Figure 5.19: Annual mean concentrations of sulphate (in µg/m^3 as S) as computed by the Euro-Delta models for the German EMEP monitoring stations. Observational data is printed in black. The EMEP model is Model04 printed in green.

Comparisons have been started to explore differences in model responses towards changes in emissions. As an example, Figure 5.20 presents for a number of European regions changes in annual mean PM_{10} concentrations (calculated from daily model results) for a number of emission control scenarios. The x-axis lists the various regions in Europe (00=Europe, 01=Austria, 08=France, 09=Germany, 12=Italy, 14=Netherlands, 19=Spain, 22=British Isles. The lines indicate the range of model results (green=highest result of all participating models, blue=lowest result, red=ensemble model, calculated from all models as the median of the daily results). The first two panels provide annual mean PM_{10} for the emissions of 2000 and CLE2010. The others indicate the percentage changes in relation to the values of 2000 or CLE for the various emission control cases (CLE, NO_x-MFR, VOC-MFR, NO_x+VOC-MFR, as well as for the ensemble model the difference between the joint NO_x/VOC case and the sums of the individual NO_x and VOC changes (i.e., the error from a linearity assumption). In most cases, the response of the EMEP model is close to the ensemble model.

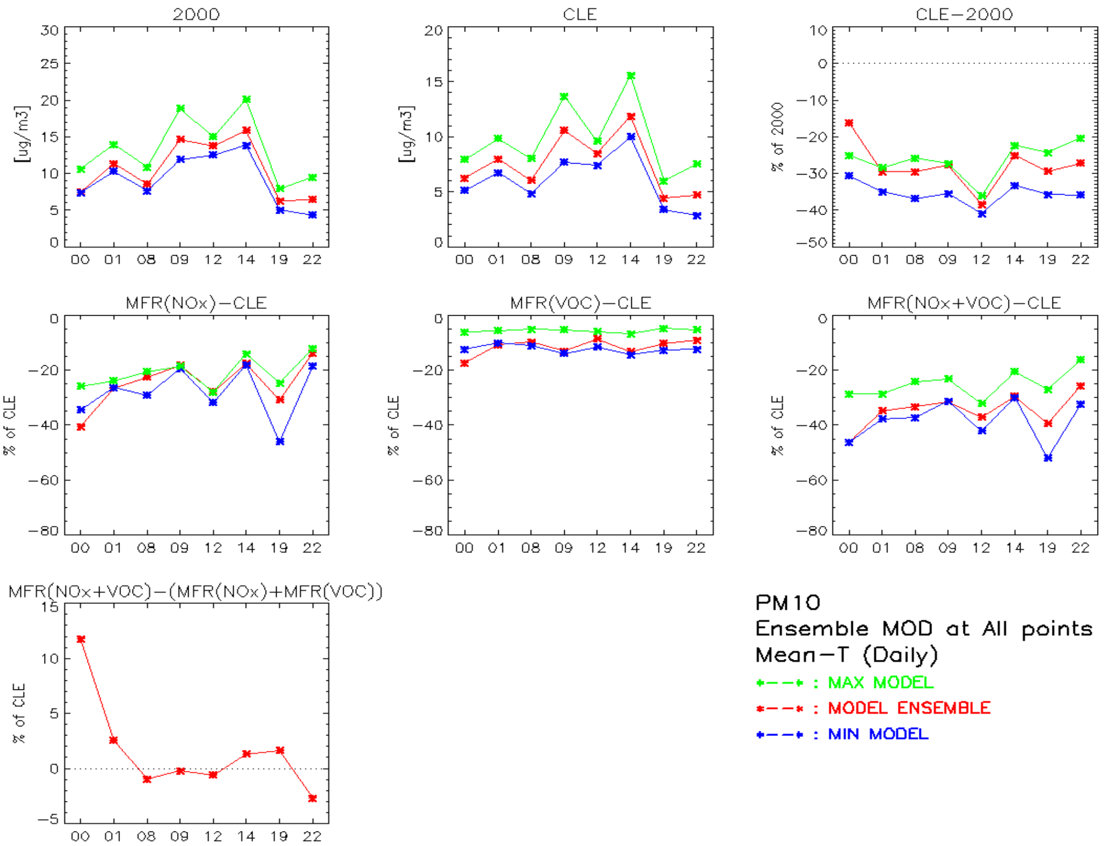

Figure 5.20: Responses of European scale dispersion models to changes in precursor emissions for different regions in Europe. Details are given in the text.

Work is in progress to further analyse these findings and to draw conclusions that can be used for the uncertainty assessment in RAINS. Emphasis will be placed on the uncertainty range spanned by these state-of-the-art models, the performance of the ensemble model that in many cases shows the closest agreement with available monitoring data, the position of the EMEP model within this uncertainty range and potential biases resulting from the choice of any particular model for the policy analysis.

5.5 State of progress and plans for further work

The description provided above summarizes the present state of work and the conceptual thinking for steps to conclude the analysis. Due to late delivery of important input information (e.g., EMEP model results, City-Delta and Euro-Delta results, etc.), the work is not yet completed. Next steps include:

- determining the linearity of regional scale dispersion of fine particles,
- constructing appropriate regional-scale source-receptor relationships,
- developing and implementing the urban module of RAINS,
- bringing the Euro-Delta exercise to a conclusion and drawing the lessons for the uncertainty analysis,
- conducting the sensitivity cases suggested by the review of the EN/ECE-WHO Task Force on Health with the final model set-up, and
- assessing the overall uncertainties of the health impact assessment.

5.6 References

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6 Modelling of health impacts of ground-level ozone

6.1 Health impacts of ground-level ozone

Back in 1999, policy analysis with RAINS for the NEC Directive and the Gothenburg Protocol relied on the health guidelines of the World Health Organization for Europe, which specify a guideline value of 60 ppb as an eight hour average (WHO, 2000). At that time, the guideline value was considered as a threshold, below which no significant health effects could be detected, but no quantification of the effects of higher concentrations was available. Consequently, the RAINS model used an AOT60 (i.e., the accumulated excess concentrations over a threshold of 60 ppb) as a proxy for quantifying exceedances of the guideline value as a measure on the way towards the no-effect level (Amann and Lutz, 2000). With this approach, no judgement was assumed on the relative importance of a large one-time excess of the 60 ppb threshold compared to repeated small violations.

In 2003, the WHO systematic review of health aspects of air quality in Europe concluded that since the time these guidelines were agreed, there is sufficient evidence for their reconsideration. The review found that recent epidemiological studies have strengthened the evidence that there are short-term O_3 effects on mortality and respiratory morbidity and provided further information on exposure-response relationships and effect modification. There is new epidemiological evidence on long-term O_3 effects and experimental evidence on lung damage and inflammatory responses. There is also new information on the relationship between fixed site ambient monitors and personal exposure, which affects the interpretation of epidemiological results.

The UN/ECE-WHO Task Force on Health "*noted that the AOT60 concept used previously within the RAINS model might no longer be appropriate to account for the effects of ozone on human health in the light of the findings of the review published by the WHO/ECEH Bonn Office. In particular, the WHO review had concluded that effects might occur at levels below 60 ppb, which was the threshold level used to calculate AOT60, and a possible threshold, if any, might be close to background levels and not determinable. This review had also indicated that the effects of ozone on mortality and some morbidity outcomes were independent of those of PM*" (TFH, 2003).

The Task Force "*invited IIASA to propose a methodology to include the effects of ozone on mortality into integrated assessment modelling. Such a methodology should:*

- *Allow for calculations of attributable deaths, based on information from a meta-analysis of time-series studies. Mr. R. Anderson (United Kingdom) informed the Task Force that such a meta analysis was currently being conducted by St. George's Hospital in London, as part of the WHO/ECEH systematic review project. He explained that the meta-analysis would make use of an extensive database of time-series studies which was continuously updated at St. George's Hospital;*
- *Base its exposure assessment on urban background ozone concentrations for urban populations (mean of daily eight-hour maximum values);*
- *Be robust in relation to key assumptions. In particular, IIASA was requested to investigate the influence of hemispheric ozone background concentrations on the selected approach;*

Include in the sensitivity analysis the study of consequences of limiting the analysis of impacts to the summer season using the summer-specific relative risk coefficients."

Due to the late availability of the new Eulerian EMEP model, only limited progress was made to date to develop a methodology for assessing mortality impacts from ozone. It is planned that the analysis would use a health-relevant metric of ozone to calculate the attributable deaths based on the findings of the time series studies. This is a major difference to the PM assessment, for which RAINS relies on cohort studies and estimates the loss in life expectancy. As shown by Rabl (2003), these two metrics cannot be directly related to each other, which will hamper the interpretation of the RAINS results.

The precise method of the available time series studies will be a major determinant for the design of the RAINS ozone health module. From the health side, the WHO review found effects on days with ozone below the former 60 ppb threshold, but could not positively identify another threshold level.

In the extreme case, with this information an assessment could thus operate without any threshold, and relate health impacts to mean ozone concentrations down to zero. Such an approach implies that any ozone molecule hitting a human has to be considered potentially harmful and, if used as a guidance for emission control strategies, such a concept would ask for reducing ozone levels below natural background.

While an integrated assessment must insist on further evidence before embarking on such extreme interpretations, attention must also be paid to the performance of the available modelling tools for different ozone regimes. The findings of the City-Delta model intercomparison shed serious doubts on the performance of the state-of-the-art dispersion models for low ozone situations. As an illustration, Figure 6.1 presents two frequency plots of hourly ozone concentrations as calculated by the models participating in the City-Delta exercise for two urban stations in Berlin (Neukölln and Buch), for the range from 0 to 60 ppb for the full year. Obviously, there is little match with the actual observations (the black line), which is not necessarily too surprising, since in all earlier model analyses the focus was on reaching good performance for high ozone situations. Thus, all models struggle to reproduce low ozone regimes, especially in the winter. As shown in Figure 6.2, the performance improves considerably for the summer half year, although the precision for the very low concentrations remains low. Thus, a balanced assessment needs to judge the strength of observational evidence on health impacts against the ability of contemporary models to produce robust results over the full range of ozone levels. A decision on this subject might involve a subjective judgment of the RAINS modellers for a particular threshold to be applied. Unfortunately, neither the meta-analysis of the time series health studies commissioned by WHO nor the final model runs of the dispersion models are finalized to date, so that no definite conclusion can be drawn at the moment. Given the low performance of ozone models for the winter season, it would be useful if the meta-analysis of the time series studies explored summer-specific relative risk coefficients.

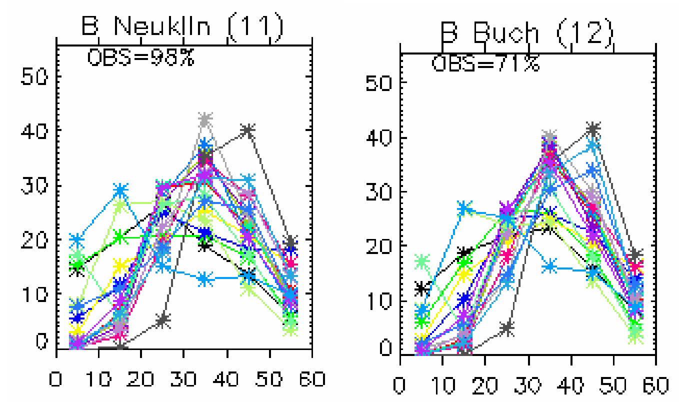

Figure 6.1: Frequency plots of 12 months hourly ozone concentrations calculated by the City-Delta models for the range from 0-60 ppb for two monitoring stations in the city area of Berlin. The black line represents the observations.

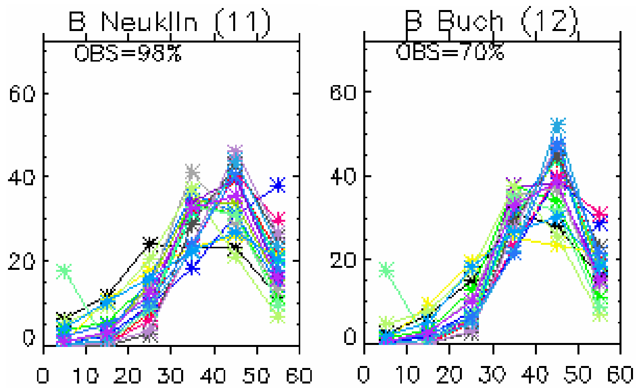

Figure 6.2: Frequency plots of summer ozone calculated by the City-Delta models for the range from 0-60 ppb for two monitoring stations in the city area of Berlin. The black line represents the observations.

6.2 Atmospheric source-receptor relationships for ground-level ozone

Although the final decision on health-relevant metrics for ozone is still outstanding, it is clear that any comprehensive assessment must consider ozone over an extended period (e.g., the summer half year) and not restrict itself to a few individual days or episodes. Thus, a preliminary analysis has been conducted to explore the ozone response of the EMEP Eulerian model towards changes in emissions. This assessment has essentially the same objectives as described in the Chapter on PM health impacts, i.e., to detect potential non-linearities that will require a different approach for the integrated assessment.

The non-linear response of ozone levels towards changes in its precursor emissions is a well-known and frequently demonstrated effect, which poses major challenges for NO_x reductions if they result in increased ozone levels. In the seventies and eighties, this non-linearity was demonstrated for peak ozone concentrations, and was usually illustrated with the help of so-called EKMA diagrams (Figure

6.3). The effect is most pronounced in areas with high NO_x emission densities, where NO_x exerts a titrating effect on ozone.

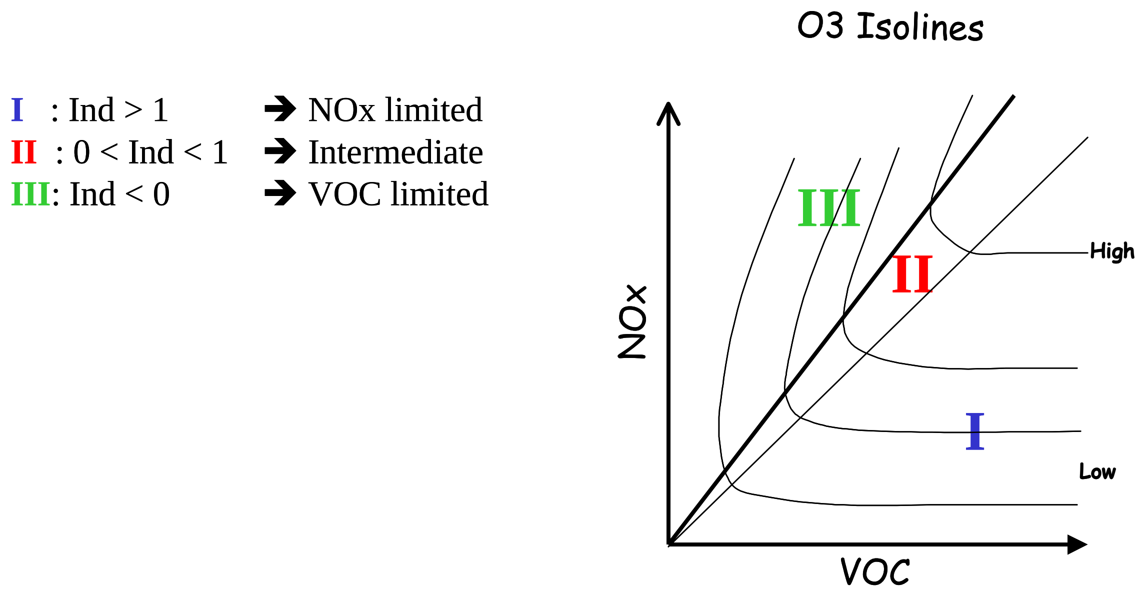

Figure 6.3: An example of an EKMA diagram illustrating the non-linearities in ozone changes towards changes in the precursor emissions

For the RAINS analysis preparing for the Gothenburg multi-pollutant/multi-effect protocol, IIASA established the validity of such non-linear ozone responses also for long-term ozone metrics, based on calculations of the EMEP Lagrangian model (Heyes *et al.*, 1996). In the subsequent statistical analysis, IIASA developed reduced-form relations in the form of quadratic polynomials that described the response of the full EMEP model towards changes in NO_x and VOC emissions with very high accuracy. This work created the basis for including ozone in a multi-pollutant/multi-effect integrated assessment, which finally found its application in the policy analysis.

Work is now underway to repeat this analysis for the new EMEP Eulerian model, where the refined 50*50 km spatial resolution should a priori result in an increased likelihood for non-linear effects around urban areas. On the other hand, if 2010 is taken as a starting year for the policy analysis, NO_x emission densities should be substantially reduced due to the measures of the NEC Directive and Gothenburg Protocol, so that the chemical regime will be different from today's situation. It remains to be analysed how these two aspects counteract each other.

It is also clear that non-linearities are strongly dependent on the specific metric considered in the analysis. Analysis suggests for decreasing emissions a continuous decline of peak ozone, while less NO_x in highly polluted urban areas, where titration is an important mechanism, will lead to (moderate) increases in the very low ozone levels. Thus, the overall effect is very sensitive to a selected threshold for the analysis, and the discussion about the validity and practicalities of health-relevant thresholds must not ignore this aspect.

The analysis is further complicated by the different spatial scales that need to be applied for the assessments of health impacts of ground-level ozone. While non-linearities occurred at least in the past for regional ozone calculations, the extended scope to cities requires additional investigation of urban areas.

6.2.1 Regional-scale source-receptor relationships

Late in 2003, the EMEP Eulerian model for ozone was finalized and reviewed by the UN/ECE Task Force for Modelling and Monitoring (TFMM, 2003). The Task Force agreed that *"There was a high level of confidence in the EMEP model's representation of the broad spatial pattern of ozone exposure levels across Europe and of the major areas of VOC and NOx limitation. This level of confidence extended to the assessment of the exposure levels required to estimate ozone crop and vegetation impacts on the regional scale and to the regional background levels, which are an essential input to the estimation of health impacts on the urban scale. However, there was limited confidence in the model's ability to evaluate more advanced strategies, for example, for individual emission source categories because of the limited VOC speciation provided in European emission inventory data and the necessarily simplified chemical mechanism employed."*

To explore the response of the EMEP model towards changes in precursor emissions, the same 87 model experiments with the EMEP Eulerian model as described in the PM chapter have been performed and various metrics of long-term ozone have been explored. Of particular interest was the detection of similar non-linearities between NO_x and VOC emission reductions as have been produced by the EMEP Lagrangian model and represented by the reduced-form model of Heyes et al., 1996.

The following graphs compare changes of calculated mean of the daily maximum ozone concentrations over the summer half year for the German grid cells (red crosses) and other European receptors (black crosses) resulting from changes in German emissions.

Figure 6.4 shows the ratio between changes in VOC emissions and resulting changes in ozone concentrations. There is an almost perfect linear relationship, both for receptors close to the sources and remote sites. As demonstrated by Figure 6.5, however, the response is dependent on the overall pollution level, in particular on the level of NO_x emissions. There is a larger response at the UFR level with ultra-low NO_x emissions than at the CLE level depicting the expected 2010 NO_x emissions. In the reduced-form model, this effect has been reflected by a variable for "effective NO_x", i.e., the NO_x accumulated along the trajectory and arriving at a receptor point.

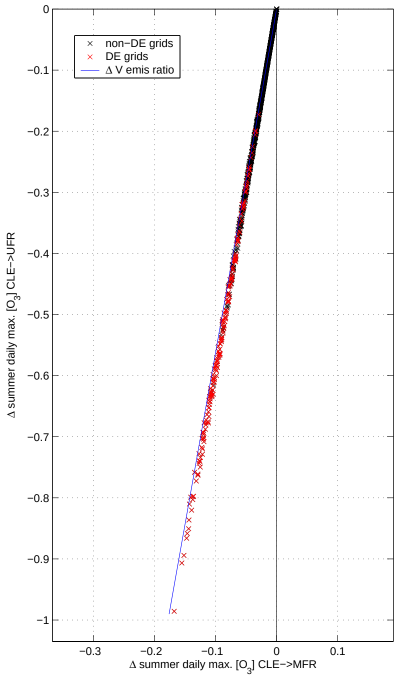

Figure 6.4:Change of mean of daily maximum ozone in the summer due to changes in German VOC emissions from CLE to MFR versus the ozone changes resulting from a reduction of German VOC emissions from CLE to UFR

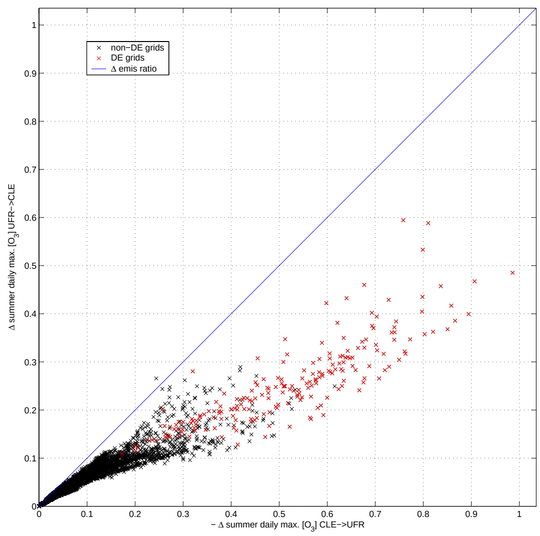

Figure 6.5: Differences in summer mean concentrations of daily maximum ozone resulting from a change of German VOC emissions from UFR to CLE with all other European emissions at UFR versus a change of the German VOC emissions from CLE to UFR with all other European emissions at CLE

As expected, a more complex situation arises for changes in NO_x emissions, which shows a clear deviation from linearity (Figure 1.6). However, at least the ratios between the deltas of the more distant sites (black crosses) are linear, suggesting a quadratic dependency as implemented in the earlier Heyes reduced-form ozone model for RAINS. Sites close to emission sources in Germany (the red crosses) show an even more complex response, and some grid cells even an increase in ozone. Furthermore, there is a small influence of the overall pollution level on the response at distant stations, while the response of the near-by German sites does not depend on the level of other pollutants (Figure 1.7). Further work will be necessary to explore this behaviour in more detail and to develop an appropriate mathematical description that can be used for the RAINS optimisation analysis.

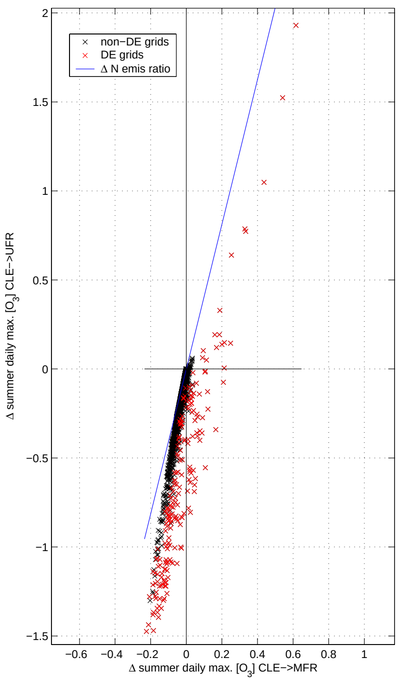

Figure 6.6: Change of mean of daily maximum ozone in the summer due to changes in German NO_x emissions from CLE to MFR versus the ozone changes resulting from a reduction of German NO_x emissions from CLE to UFR

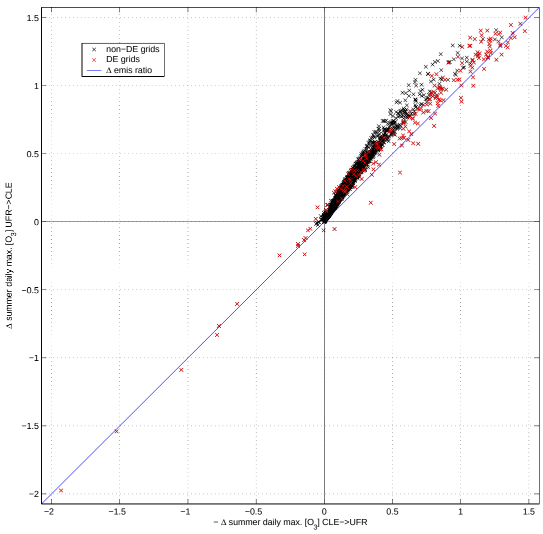

Figure 6.7: Differences in summer mean concentrations of daily maximum ozone resulting from a change of German NO_x, emissions from UFR to CLE with all other European emissions at UFR versus a change of the German NO_x emissions from CLE to UFR with all other European emissions at CLE

6.2.2 Urban-scale source-receptor relationships

Since a large share of the European population is living in cities, a health impact assessment needs to address ozone in urban areas. In principle, RAINS uses for ozone the same concept as for fine particles: the regional scale ozone formation will be modelled with the EMEP Eulerian model, and it is envisaged to derive a sufficiently simple formulation that allows the estimation of ozone in urban areas from the regional scale (background) calculations with a limited set of city-specific information.

For this purpose, IIASA has initiated the City-Delta project (http://rea.ei.jrc.it/netshare/thunis/citydelta), which brought 20 modelling teams together to understand the systematic differences between regional scale and urban scale model results. More information on this activity is provided in the PM chapter.

To date, 340 sets of six-month calculations have been received by JRC. It is mentioned above that all models share difficulties in reproducing very low ozone concentrations, especially in the winter time. Another important finding is that models with finer resolution, if compared to measurements within cities, do not perform significantly better than the 50 km models. This might be surprising on the basis of the more detailed modelling of chemical processes of fine scale models. In practice, however, the theoretically superior performance of such models seems to be counteracted by increased uncertainties and inaccuracies in emission data (e.g., with a 1*1 km resolution on an hourly basis over the full year) or meteorological input, which is in many cases not available with the same fine spatial and temporal resolution of the models. For the integrated assessment, these findings might indicate some “natural” limits to the accuracy in ozone modelling that can be achieved with the presently available tools.

Figure 6.8 provides a summary plot of the changes in urban ozone levels towards changes in precursor emissions, as calculated by the models participating in City-Delta 1. While there is some divergence in the predictions of the changes between 1999 and CLE 2010, there is striking consistency in the direction and magnitude of further changes beyond 2010. It is also interesting to note for the post-2010 situation, the responses of regional dispersion models (left panel) and urban models (right panel) are in most cases strikingly similar, even for diverse cities such as Milan and Paris. The figure also reveals the persistence of non-linear ozone responses beyond the year 2010, at least for some European cities.

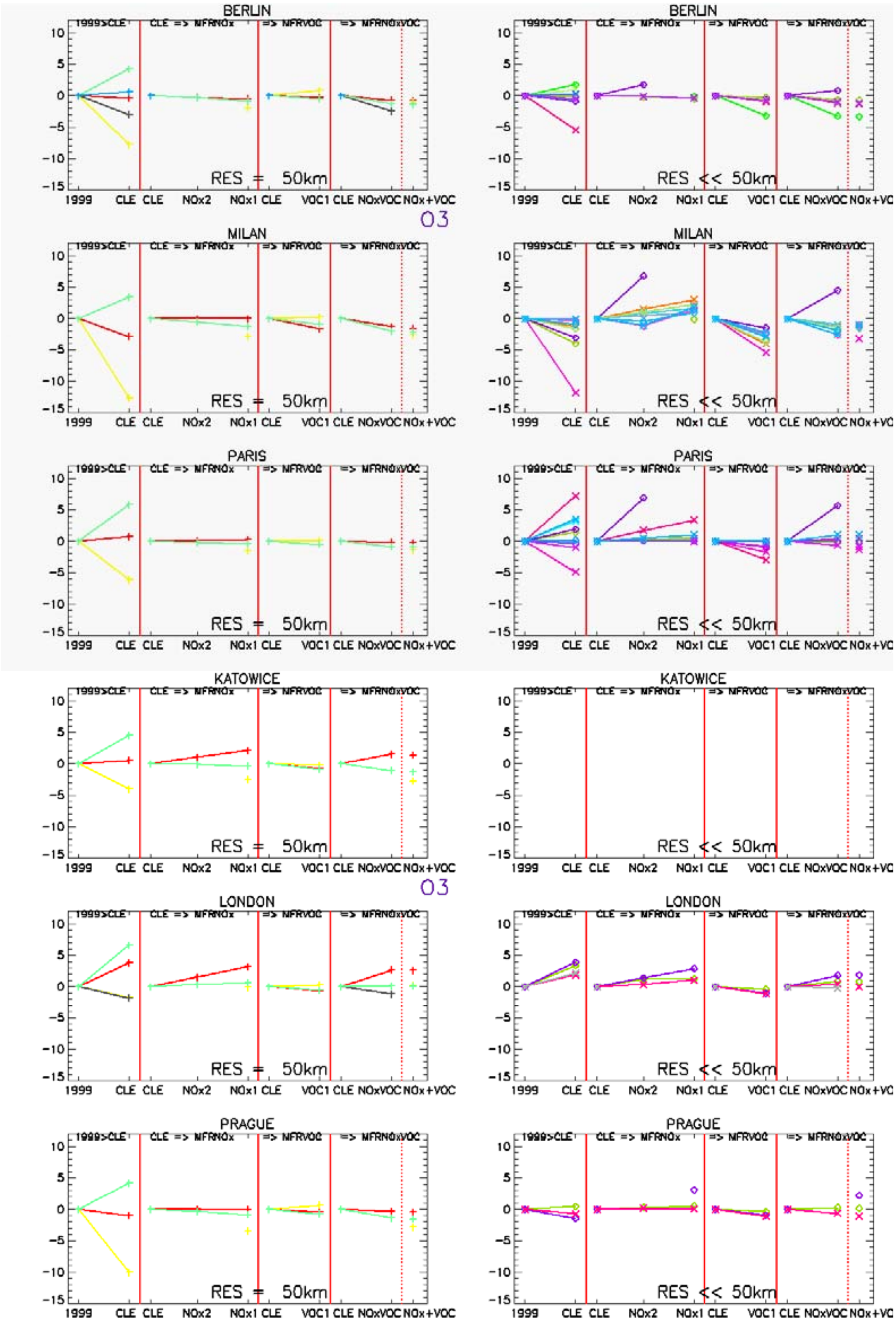

Figure 6.8: Summary of City-Delta Phase 1 model responses of urban ozone levels (summer mean concentrations) towards changes in the various precursor emissions for six cities. The left panel shows responses of regional scale models (50 km resolution), while the right panel shows responses of fine scale models. Four "deltas" are presented: (1) the change between emissions of 1999 and CLE 2010; (2) the change from CLE 2010 to MFR NO_x; (3) the change from CLE 2010 emissions to MFR for VOC, and (4) the change from CLE2010 to MFR for NO_x and VOC. Different models are displayed in different colours.

An in-depth analysis has been conducted to explore the implications of different resolutions on health-relevant output. Figure 6.9 and Figure 6.10 present the spatial pattern of ozone changes calculated for Berlin for the emission changes between 1999 and 2010 by the urban and regional scale models, respectively. For Figure 6.11, the gridded ozone concentrations computed by fine and large-scale models for their model domains have been multiplied with the respective population densities. It is surprising that there are certain differences in this calculated population exposure between the models, but there is no systematic difference between the result of regional scale and urban scale models. Furthermore, the changes between the 1999 and 2010 situations as calculated by the models are in all cases smaller than the differences between the models.

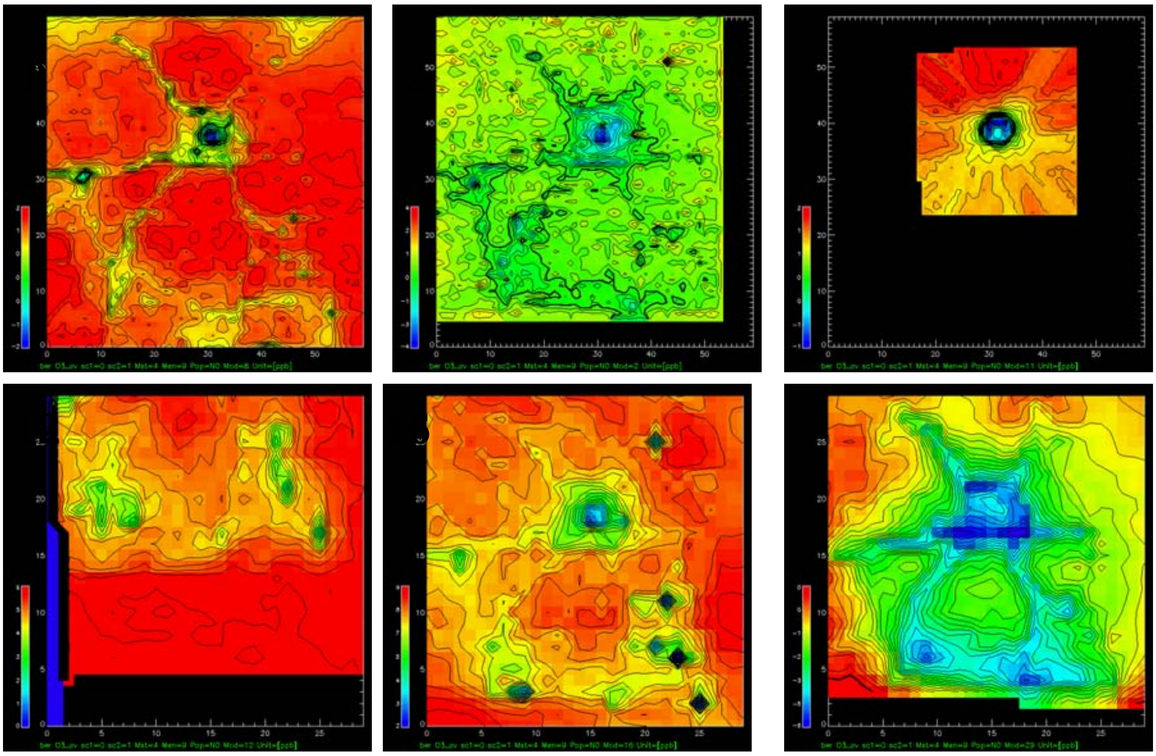

Figure 6.9: Differences in mean summer ozone between the emissions of 1999 and 2010 as calculated by urban scale models for Berlin

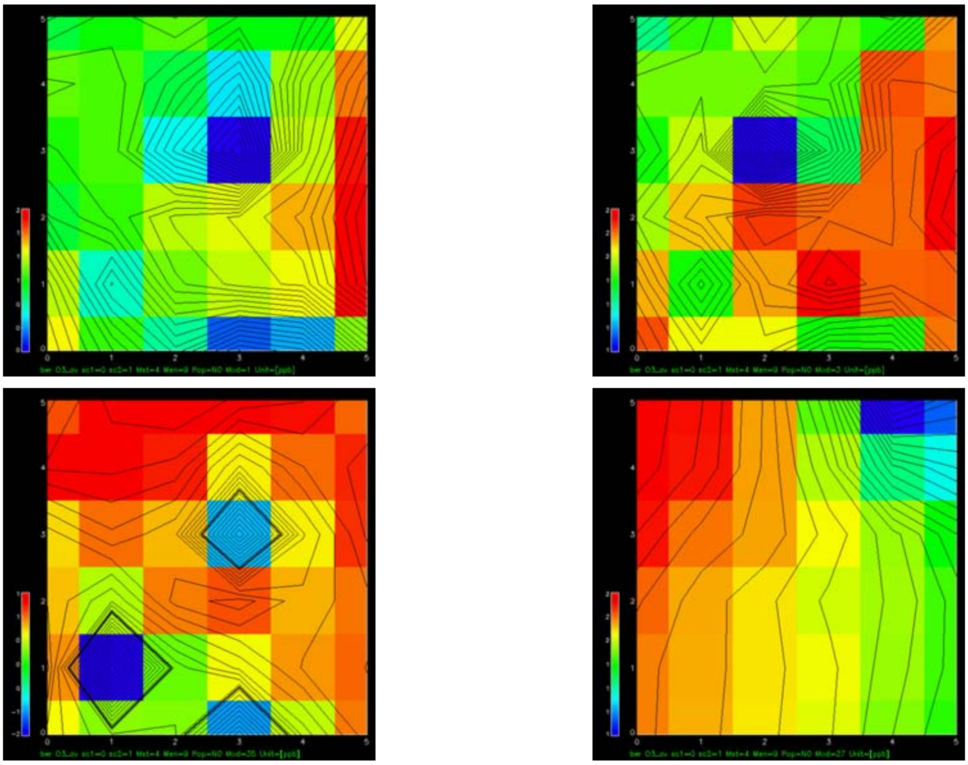

Figure 6.10: Differences in mean summer ozone between the emissions of 1999 and 2010 as calculated by regional scale models for Berlin. The model domain is the same as in the preceding graph.

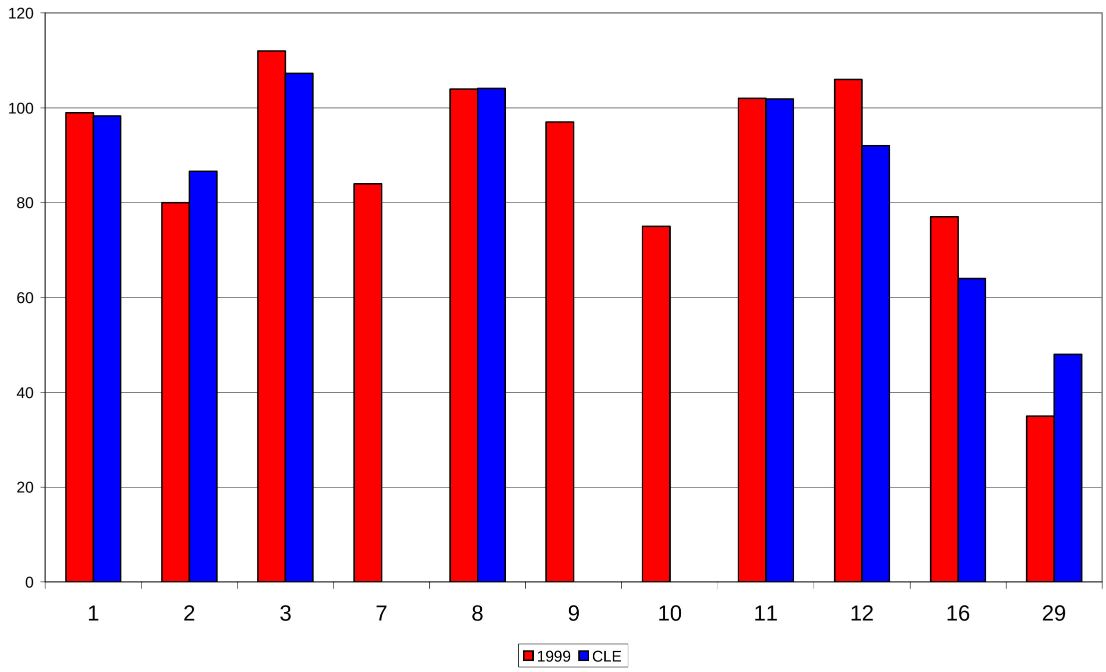

Figure 6.11: Cumulative population exposure (summer mean ozone * population density) calculated on a fine scale basis for Berlin, for 1999 and the 2010 CLE scenario

Further analysis will assess the performance and response of the model ensembles, which will provide valuable further information for the integration into RAINS. Another important aspect will be the relative performance and position of the EMEP Eulerian model within the ensemble, so that insights into potential biases of an analysis based on the EMEP model could be derived.

6.2.3 Modelling urban ozone in RAINS

For modelling urban ozone into RAINS for the purposes of health impact assessment, a similar approach as for PM is envisaged. Thereby, the regional-scale source-receptor relationships will be derived from the EMEP model, which will then deliver for an emission scenario the rural background concentrations of ozone for the grid cell where a city is located. A further step will then modify these rural concentrations to reflect the population-related characteristic ozone exposure. Initial analysis from the City-Delta emission and monitoring data reveals a striking relationship of the difference between rural and urban ozone levels and NO_x emission densities. Further work will be conducted to further explore this aspect, to include the location of population within the city and to implement it within the RAINS analysis.

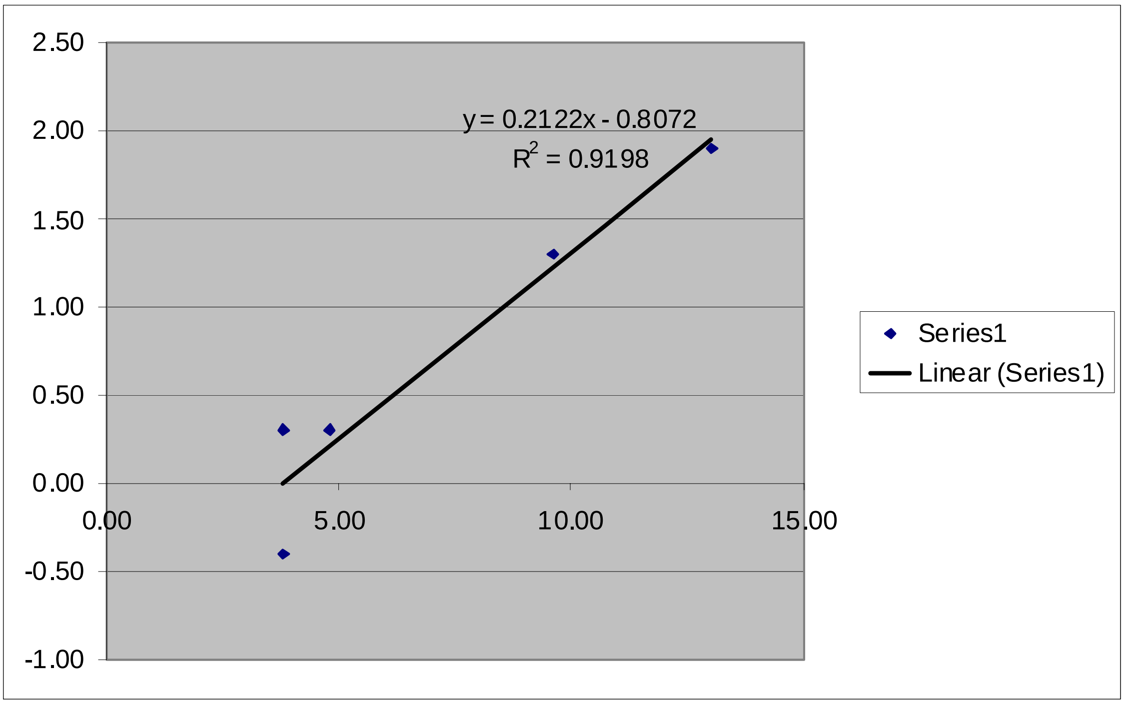

Figure 6.12: Decrease in urban long-term (summer mean) ozone compared to rural background level (in ppb) as a function of changes in NO_x emission densities in the urban model domain (t/km^2) for five City-Delta cities (Berlin, London, Milan, Paris, Prague). This graph is derived from the City-Delta ensemble solutions for the CLE and NO_x-MFR scenarios.

6.3 Uncertainties

As explained above, many aspects load any estimate of health impacts of ozone with significant uncertainties. For quantification of the health-relevant air quality changes resulting from emission changes, the general imperfections of dispersion modelling for ozone cannot be eliminated in the near future.

Thus, it is even more important to design the integrated assessment model system in such a way as to minimize the potential influence of the unavoidable uncertainties and maximize the robustness of model results. A key element in this task will be the choice of the appropriate ozone metric that will be used for the health impact assessment.

While the specific approach for uncertainty treatment within the integrated assessment model can only be designed once the model approach has been ultimately decided (i.e., after all results from the EMEP dispersion model and City-Delta are finally available), preparatory actions have been taken to derive quantified estimates of the uncertainties of the various elements in the model chain. City-Delta by its design provides *inter alia* information about the extent of agreement and disagreement among the available state-of-the-art urban dispersion models.

6.3.1 The Euro-Delta project

To gain insight into the performance of regional scale models and obtain an overall feeling of present uncertainties of the state-of-the-art dispersion models for ozone, IIASA together with the Institute for Environment and Sustainability of the Joint Research Centre (Ispra), MET.NO, EUROTRAC-2 and CONCAWE, has initiated the Euro-Delta model intercomparison (http://rea.ei.jrc.it/netshare/thunis/eurodelta/). The aim of this exercise is to conduct a systematic comparison of regional scale dispersion models to judge the performance of state-of-the-art regional scale dispersion models in relation to health- and policy-relevant model output.

Five European scale dispersion models including the EMEP Eulerian model participate in this intercomparison (Table 5.5), which analyses model responses for PM and ozone for seven emission control scenarios. More detail on the set-up is given in the health PM Chapter.

Table 6.1: Participating models in Euro-Delta

Model	Contact person	Affiliation
LOTOS	P. Builtjes	TNO-MEP, (NL)
REM3/CALGRID	R. Stern	FUB, (D)
CHIMERE	C. Honore L. Rouil	INERIS, (F)
Unified EMEP	L. Tarrason	EMEP/MSC-W, (N)
MATCH	J. Langner	SMHI (S)
MODELS-3	I. Rodgers	INNOGY, (GB)

Figure 6.13 compares the summer mean ozone concentrations calculated with the Euro-Delta models with observations for German EMEP monitoring sites.

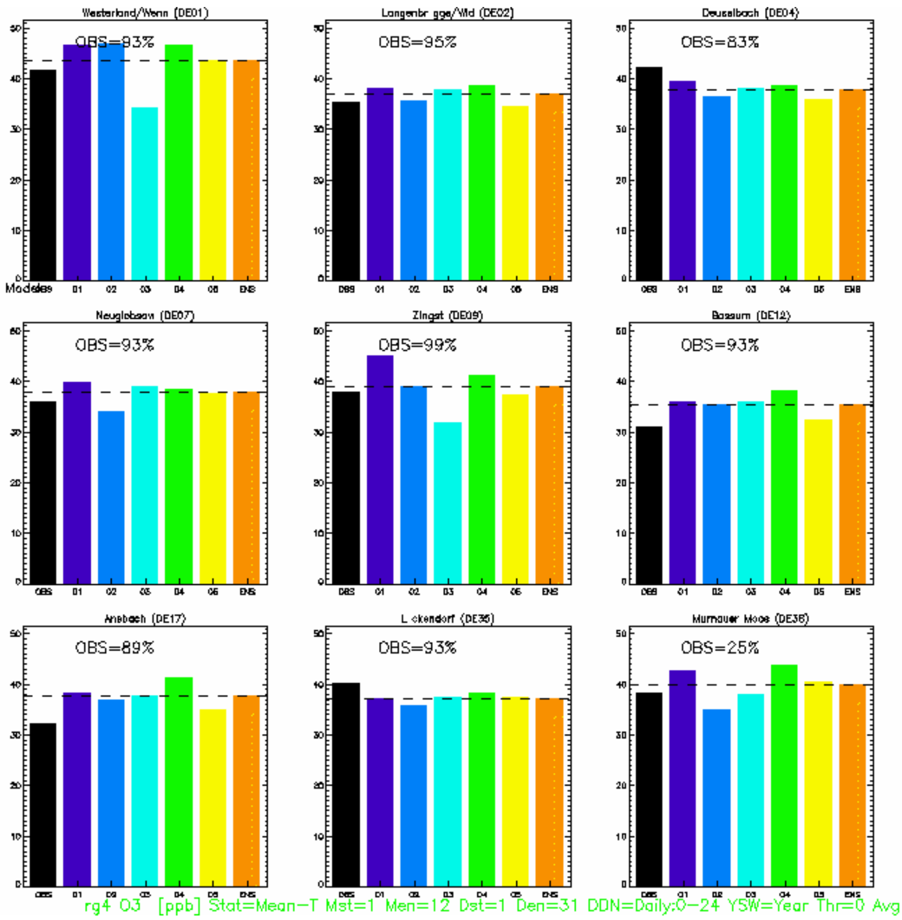

Figure 6.13: Summer mean ozone concentrations computed by the Euro-Delta models for the German monitoring stations (in ppb). The black bar indicates observations

Comparisons have been started to explore differences in model responses towards changes in emissions. As an example, Figure 6.14 presents for a number of European regions changes in summer mean ozone concentrations (calculated from daily model results) for a number of emission control scenarios. The x-axis lists the various regions in Europe (00=Europe, 01=Austria, 08=France, 09=Germany, 12=Italy, 14=Netherlands, 19=Spain, 22=British Isles. The lines indicate the range of model results (green=highest result of all participating models, blue=lowest result, red=ensemble model, calculated from all models as the median of the daily results). The first two panels provide summer mean ozone for the emissions of 2000 and CLE2010. The others indicate the percentage changes in relation to the values of 2000 or CLE for the various emission control cases (CLE, NO_x-MFR, VOC-MFR, NO_x+VOC-MFR, as well as for the ensemble model the difference between the joint NO_x/VOC case and the sums of the individual NO_x and VOC changes (i.e., the error from a linearity assumption). In most cases, the response of the EMEP model is close to the ensemble model.

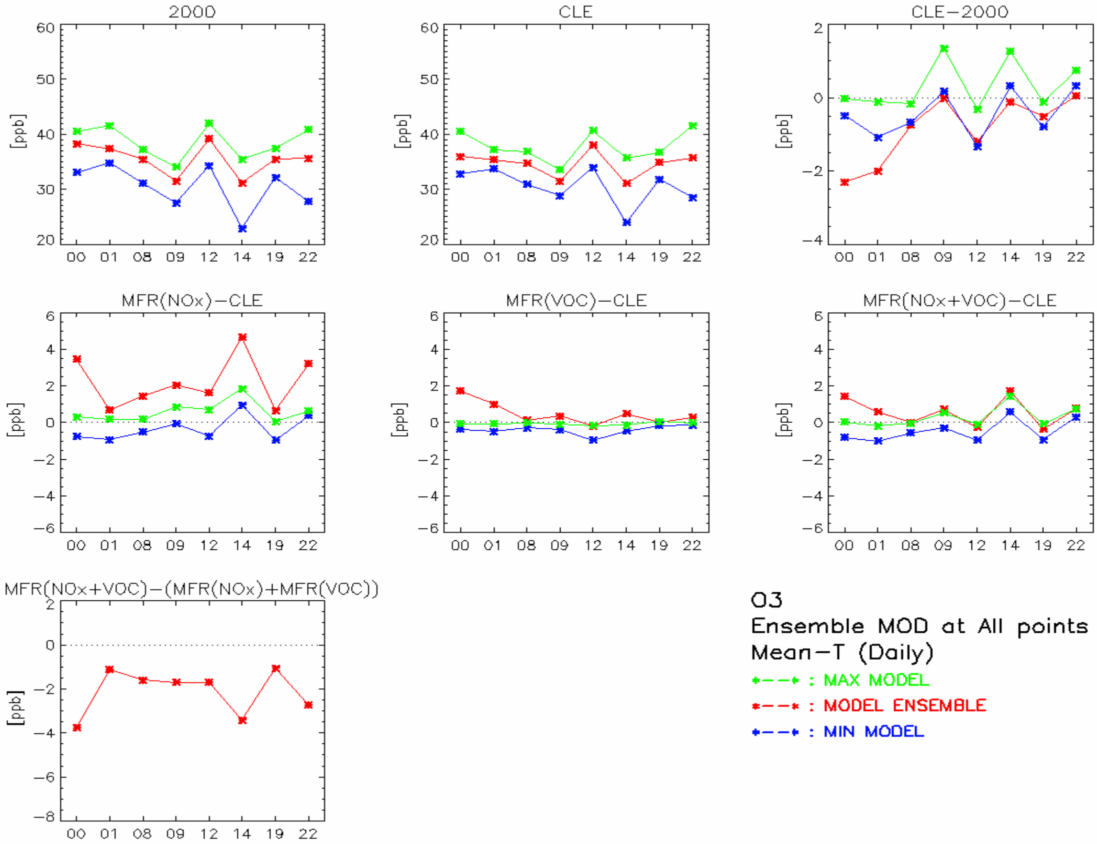

Figure 6.14: Responses of European scale dispersion models to changes in precursor emissions for different regions in Europe. Details are given in the text.

6.4 State of progress and plans for further work

At the moment, IIASA has received the first 87 model experiments from the new EMEP Eulerian model and started an in-depth analysis of the model behaviour. Next steps include:

determining the linearity of regional scale dispersion of ozone within the given emission constraints,

constructing appropriate regional-scale source-receptor relationships,

developing and implementing the urban module of RAINS,

bringing the Euro-Delta exercise to a conclusions and draw the lessons for the uncertainty analysis,

designing and implementing the health impact assessment for ozone as suggested by the UN/ECE-WHO Task Force on Health with the final model set-up, and

- assess the overall uncertainties of the health impact assessment.

6.5 References

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7 Modelling of vegetation impacts of ground-level ozone

7.1 Vegetation impacts of ground-level ozone

For the policy analysis for the NEC Directive and the Gothenburg Protocol in 1999, the RAINS model applied the concept of critical levels to quantify progress towards the environmental long-term target of full protection of vegetation from ozone damage. For vegetation, the UN/ECE Working Group on Effects expressed in its Mapping Manual the critical levels for crops, forests and semi-natural vegetation in terms of different levels of AOT40, measured over different time spans. While, following the advice of the mapping community, the RAINS model has never used the AOT40 approach for quantifying vegetation damage, it used the AOT40 to put a figure on the progress towards the environmental long-term targets. These figures were used as indicators in the scenario analysis to compare the vegetation impacts of alternative emission control scenarios in relative terms, and they were used to provide a scale for the environmental interim objectives for the optimisation analysis.

In the following years, several important limitations and uncertainties of the AOT approach have been recognised. In particular, the real impacts of ozone depend on the amount of ozone which reaches the sites of damage within the leaf, whereas the concentration-based Critical Level only considers the ozone concentration at the top of the vegetation canopy. Alternative concepts, including the ozone flux concept, were developed and suggested as superior alternatives to replace the former AOT40 approach.

In 2002, a workshop on *Establishing Ozone Critical Levels II* was held in Gothenburg (Karlsson *et al.*, 2003a) to review the scientific findings made in different areas relevant for a level II approach and to recommend the further steps towards practical implementation of a more refined approach.

The conclusions of this workshop were reported along the reporting lines of the UN/ECE Working Group on Effects, so that on 10-13 February 2004 a draft for a revised mapping manual has been presented to the ICP Vegetation Task Force.

To any reasonable extent, the integrated assessment of RAINS attempts to base its analysis on peer-reviewed and formally approved scientific approaches to ease political acceptance of the resulting RAINS model outcomes. Thus, the recommendations specified in the mapping manual of the UN/ECE Working Group on Effects is of direct relevance to the approach taken by RAINS.

The draft mapping manual in its version of February 10, 2004 (UN/ECE, 2004) specifies for ozone concentration-based Critical Levels and flux-based Critical Levels, using different scientific bases of risk assessment. Both of these approaches incorporate the concept that the effects of ozone are cumulative, and therefore that risk assessment must incorporate a summation of ozone concentrations, or instantaneous fluxes, over the growing season of the vegetation. In some cases, it is recommended that only part of the growing season, representing the stages which are most sensitive to the effects of ozone, should be used. In all cases, a threshold concentration or flux is used, and only concentrations or fluxes above this threshold are incorporated into the risk assessment.

The first approach is to use the concentration accumulated over a threshold concentration during daylight hours over the appropriate time window (based on the growing season of the receptor). This value is expressed in units of ppb h, ppm h, or nmol mol-1 h. The term AOTX (Concentration Accumulated over a Threshold Ozone Concentration of X ppb) has been adopted for this index. The

AOT40 and the AOT30 indices (using thresholds of 40 ppb and 30 ppb respectively) are used to define Critical Levels in the mapping manual. These Concentration-based Critical Levels of ozone, CLec, are the values above which direct effects of ozone may occur according to current knowledge.

The second approach is to use the stomatal flux of ozone, F_{st} (in nmol m^{-2} PLA s^{-1}), based on projected leaf area and accumulated over a stomatal flux threshold of Y nmol m^{-2} s^{-1}, over the appropriate time window based on the growing season ($AF_{st}Y$). This Accumulated Stomatal Flux of Ozone above a Flux Threshold ($AF_{st}Y$), is calculated as the sum over time of the differences between instantaneous or hourly values of F_{st} and Y nmol m^{-2} PLA s^{-1} for the periods when F_{st} exceeds Y. The Flux-based Critical Level of ozone, CLe_{flux} mmol m^{-2} PLA, is then the cumulative stomatal flux of ozone, $AF_{st}Y$, above which direct adverse effects may occur according to present knowledge. Values of CLe_{flux} have been identified for crops and forest trees, but this approach cannot yet be applied to semi-natural vegetation.

7.2 Proposed critical levels

The mapping manual specifies critical levels for both approaches, though flux-based critical levels only for wheat and potato (Table 7.1, Table 7.2).

Table 7.1: Concentration-based critical levels of ozone for growth/biomass/yield changes in vegetation; Source: UN/ECE, 2004

Receptor	Time period	Critical level (AOT30, ppm h)*)	Critical level (AOT40, ppm h)
Agricultural crops	Three months	4	3
Horticultural crops	Four months	-	5
Forest trees	Growing season (six months by default)	9	5
Semi-natural vegetation	Three months (or growing season, if shorter)	-	3

*) only applicable for integrated assessment modelling

Table 7.2: Flux-based critical levels of ozone for growth/biomass/yield changes in vegetation (only available for wheat and potato); Source: UN/ECE, 2004

Receptor	Time period	Critical level (AFst6)
Wheat	900 ˚C days starting 200 ˚C days before anthesis (flowering)	1 $mmol/m^2$ projected sunlit leaf area
Potato	1130 ˚C days starting at plant emergence	5 $mmol/m^2$ projected sunlit leaf area

For implementation in RAINS, several considerations apply:

- From earlier analysis of ozone time series for various parts of Europe, the critical level for forest trees (5 ppm.hours over the full vegetation period, April 1- September 30 is recommended as default) appears as the most stringent constraint. For most parts of Europe, the other critical levels will be automatically achieved if the 5 ppm.hours over six months condition is satisfied. Thus, if used for setting environmental targets for emission reduction strategies, the critical levels for forest trees would imply protection of the other receptors. (This fact should be reconfirmed with more recent monitoring data).
- As of now, flux-based critical levels are only available for wheat and potato. While an analysis of emission control strategies would certainly benefit from a detailed assessment of crop losses of wheat and potatoes in Europe, wheat and potato losses due to ozone are not likely to emerge as the major drivers for European clean air policies. Thus, the quantification should also address other vegetation types.
- **Thus, according to current plans, RAINS aims to incorporate the critical level for forest trees as the key indicator for measuring progress in reducing vegetation damage from ozone.** Any economic evaluation, e.g., as part of the cost-benefit analysis of CAFE conducted by other teams, should however include the assessment of ozone damage to crops and semi-natural vegetation.

7.2.1 Critical levels for forest trees

At the UNECE workshop in Gothenburg in November 2002 (Karlsson *et al.*, 2003a) it was concluded that the effective ozone dose, based on the flux of ozone into the leaves through the stomatal pores, represents the most appropriate approach for setting future ozone critical levels for forest trees. However, uncertainties in the development and application of flux-based approaches to setting critical levels for forest trees are at present too large to justify their application as a standard risk assessment method at a European scale. Although $AF_{st}Y$ is much more physiologically relevant than AOTX, more time and data are needed before $AF_{st}Y$ - response relationships for trees could be considered sufficiently robust for establishing a critical load of ozone for forest trees.

Thus, the mapping manual retains the AOTX approach as the recommended method for integrated risk assessment for forest trees, until the ozone flux approach will be sufficiently refined. However, the time window over which the AOTX is accumulated should be adapted according to local phenology. The critical level value as well as the threshold value used for the AOTX, should be re-considered for certain sensitive deciduous trees species.

The experimental database that was presented at the UNECE Workshop in Gothenburg 2002 has been re-analysed and expanded to include additional correlations with AOT20, AOT30, and AOT50 (Karlsson *et al.*, 2003b). As a result, linear regressions between exposure and response have the highest r^2 values and there are no significant intercepts (Table 7.3). Using the described sensitivity categories, AOT40 gave the highest r^2 values of the AOTX indices tested. The difference between the r^2 values for AOT40 and AOT30 was, however, small.

An optional additional AOT30-based critical level of ozone has also been derived for forest trees based on the response function for birch and beech. The value for this critical level is an AOT30 of 9 ppm.h applied to the same time windows as described for AOT40. Following discussions at the Gothenburg Workshop, the 16th Task Force Meeting of the ICP Vegetation and the 19th Task Force Meeting of the ICP Modelling and Mapping, it was concluded that AOT40 remains as the main option for ozone critical levels, but that AOT30 can be used in integrated assessment modelling on the European scale if this considerably reduces uncertainty in the overall integrated assessment model.

7.3 Choice of the AOTx

Thus, the design of an integrated assessment model should consider the strength of evidence for determining the different metrics of the critical level and balance this against other aspects of model performance in the overall model context.

According to Table 7.3, in terms of the statistical quality of the fit, the AOT40 regressions outperform the AOT30 formulation. However, for birch and beech, which is used for setting the critical level, the r^2 of the AOT40 (0.62) is insignificantly higher than that for the AOT30 (0.61).

Table 7.3: Statistical data for regression analysis of the relationship between AOTX ozone exposure indices (in ppm.h) and percent reduction of total and above-ground biomass for different tree species categories. Source: UN/ECE, 2004

Ozone index/ plant category	Linear regression			
	r^2	p for the slope	p for the intercept	slope
AOT20				
Birch, beech	0.52	<0.01	0.70	- 0.357
Oak	0.57	<0.01	0.73	- 0.142
Norway spruce, Scots pine	0.73	<0.01	0.31	- 0.086
AOT30				
Birch, beech	0.61	<0.01	0.63	- 0.494
Oak	0.61	<0.01	0.79	- 0.170
Norway spruce, Scots pine	0.76	<0.01	0.61	- 0.110

AOT40				
Birch, beech	0.62	<0.01	0.31	- 0.732
Oak	0.65	<0.01	0.73	- 0.216
Norway spruce, Scots pine	0.79	<0.01	0.86	- 0.154
AOT50				
Birch, beech	0.53	<0.01	0.05	- 1.033
Oak	0.62	<0.01	0.82	- 0.248
Norway spruce, Scots pine	0.76	<0.01	0.16	- 0.188

A further factor to be taken into account is a possible difference in the performance of atmospheric dispersion models for calculating AOT30 or AOT40. As discussed in the chapter on ozone health impacts, all state-of-the-art ozone models have difficulties in accurately reproducing low ozone concentrations. This would argue for a higher threshold to increase the accuracy of ozone calculations. However, the weak performance of the available models in the winter half year is of less relevance for the vegetation impact assessment, because here only the vegetation period (April-September) need be considered.

On the other hand, the results of ozone models are strongly influenced by assumptions on the hemispheric background concentration of ozone used as boundary conditions for the continental scale calculations. Present understanding suggests hemispheric background levels between 30 and 40 ppb. While any particular choice of a specific number for the background has certain impacts on general ozone results, this uncertainty will be magnified for metrics employing a threshold, such as the AOT measure, especially if the threshold is put exactly within the uncertainty range. Thus, it can be expected that a threshold set below hemispheric background (e.g., AOT30) would deliver more robust results in terms of an AOT than a threshold put slightly above the background level (e.g., AOT40). This theoretical consideration can be demonstrated with the practical model performance of the Euro-Delta model ensemble (Figure 7.1) and the EMEP Eulerian dispersion model (Figure 7.2). In both cases calculations of the excess of the critical level expressed in AOT30 yield higher correlations with monitoring data than calculations of the AOT40 excess ozone.

While this analysis addressed the situation with present emissions, an integrated assessment also needs to consider the robustness of model calculations for the policy-relevant emission ranges, which will be the focus of the envisaged analysis. Figure 7.3 compares the changes in AOT30 (left panel) and AOT40 (right panel) for nine EMEP monitoring stations as calculated by the Euro-Delta models for a reduction of all European emissions from CLE to MFR. While the quantitative statistical analysis of this comparison has not yet been completed, a first visual inspection suggests less variability between models for the AOT30 than for the AOT40.

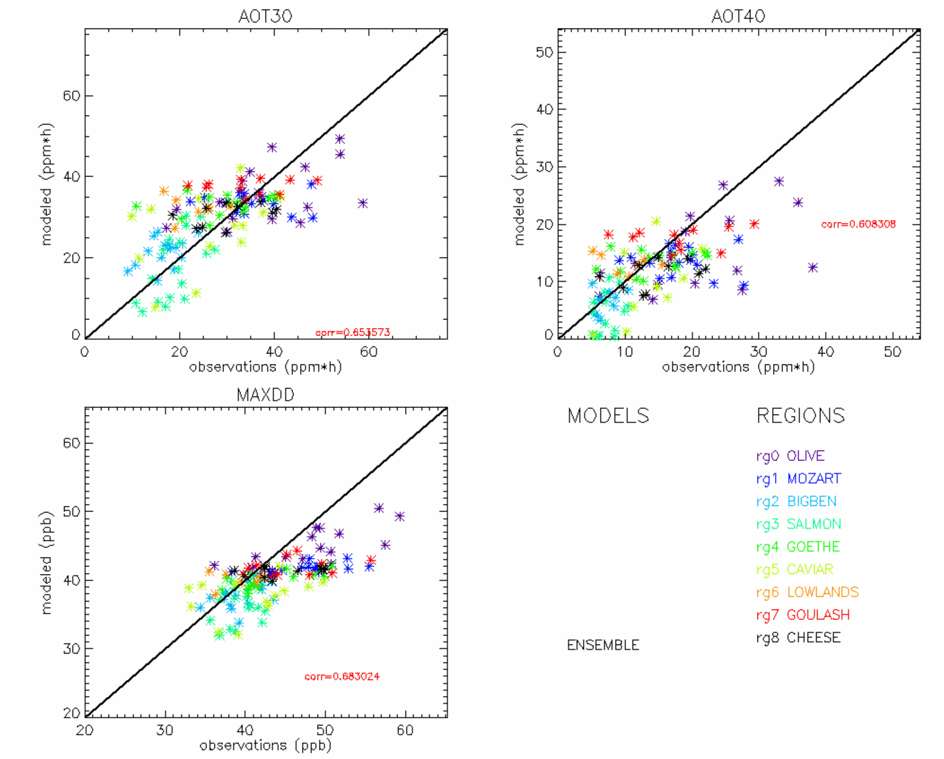

Figure 7.1: Comparison of the performance of the Euro-Delta model ensemble for the excess of the critical level expressed as AOT30 (above the critical level of 9 ppm.hours), AOT40 (above the critical level of 5 ppm.hours) and mean of daily maximum ozone concentrations, for the summer half year 1999. The data points indicate EMEP monitoring stations in the various regions of Europe. Source: Euro-Delta

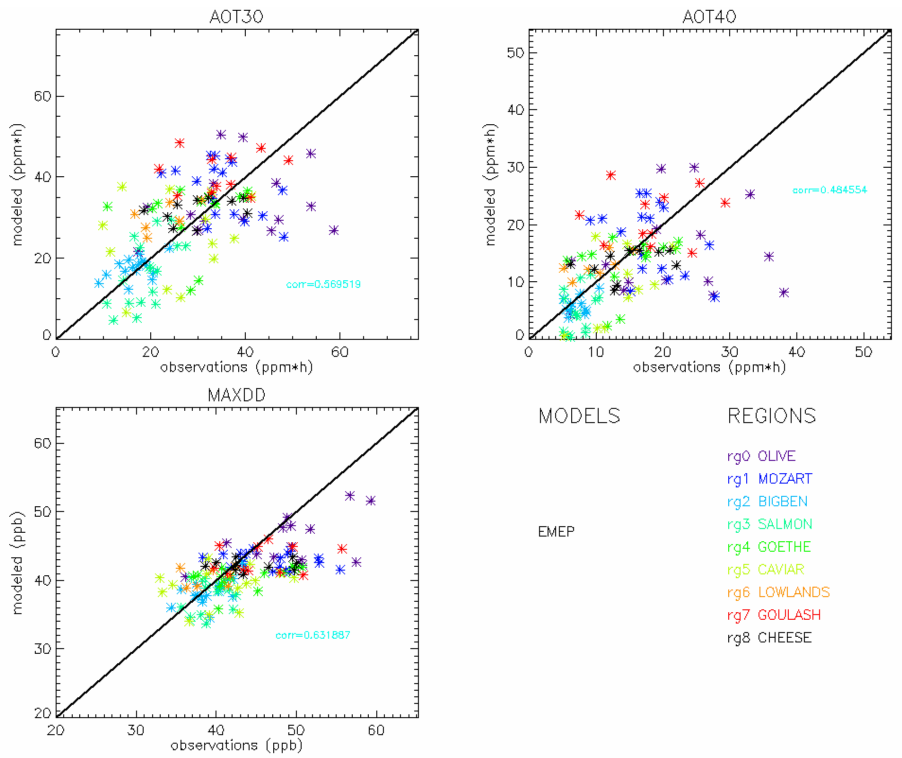

Figure 7.2: Comparison of the performance of the EMEP model for the excess of the critical level expressed as AOT30 (above the critical level of 9 ppm.hours), AOT40 (above the critical level of 5 ppm.hours) and mean of daily maximum ozone concentrations, for the summer half year 1999. The data points indicate EMEP monitoring stations in the various regions of Europe. Source: Euro-Delta

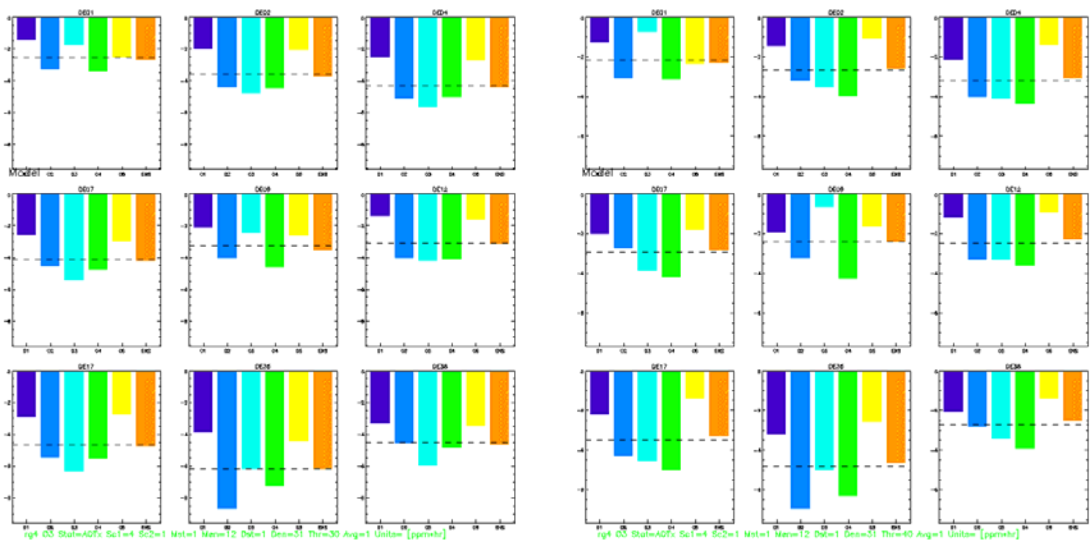

Figure 7.3: Changes in AOT30 (left panel) and AOT40 (right panel) as calculated by the Euro-Delta models (each bar presents one model) for the difference in emissions between the CLE2010 and MFR scenarios for nine German EMEP monitoring stations (in ppm.hours)

A further criterion could be the possibility to describe the response of the full EMEP Eulerian model towards changes in precursor emissions with reduced-form models. An initial assessment of the 87 EMEP model experiments has been carried out to explore the non-linearities of modelled ozone changes expressed in AOT30 and AOT40. From Figure 7.4 to Figure 7.7, both metrics produce non-linearities similar to those observed in the Lagrangian ozone model. The deviations from linearity appear more systematic for AOT30 than for AOT40, which might facilitate construction of a reduced-form model for the AOT30.

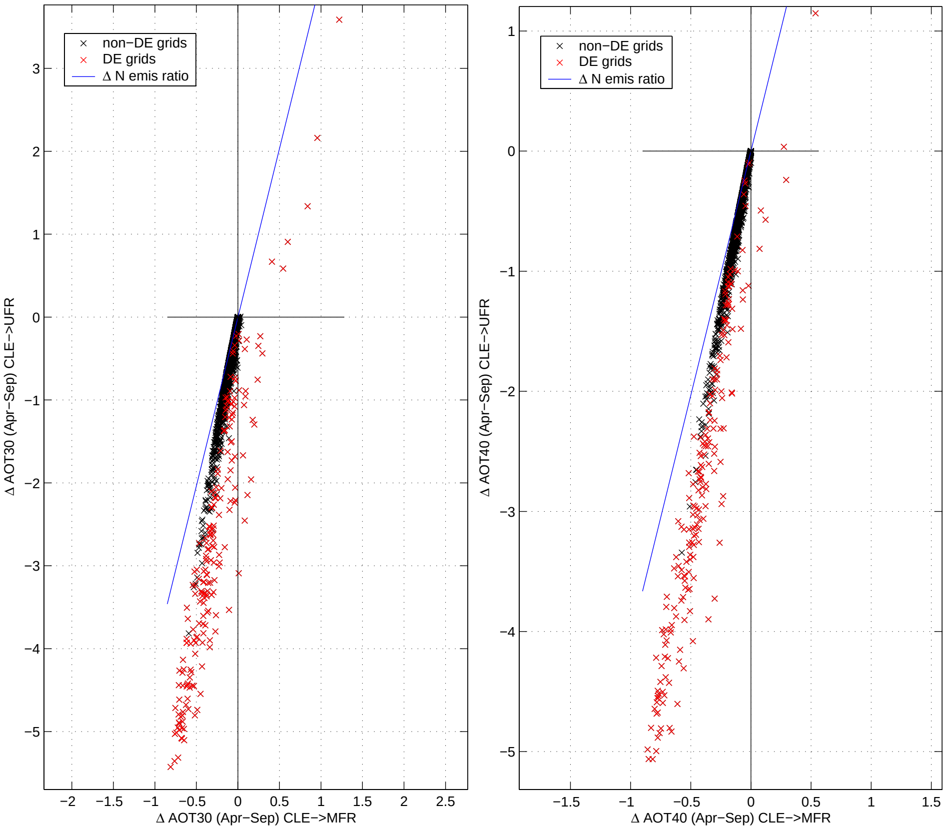

Figure 7.4: Changes in six-months AOT30 (left panel) and AOT40 (right panel) due to changes in German NO_x emissions from CLE to MFR versus the changes from a reduction of German NO_x emissions from CLE to UFR

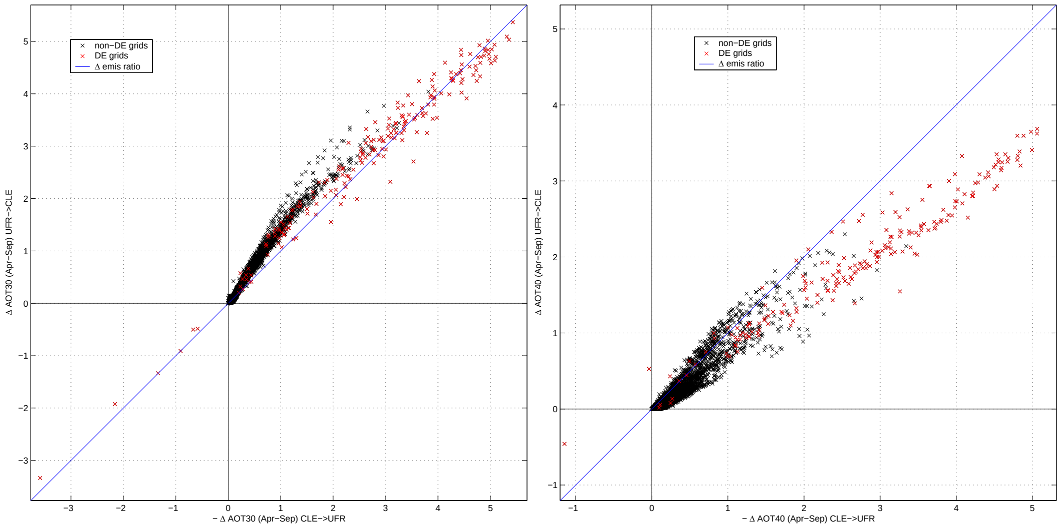

Figure 7.5: Differences in six months AOT30 (left panel) and AOT40 (right panel) resulting from a change of German NO_x, emissions from UFR to CLE with all other European emissions at UFR versus a change of the German NO_x emissions from CLE to UFR with all other European emissions at CLE

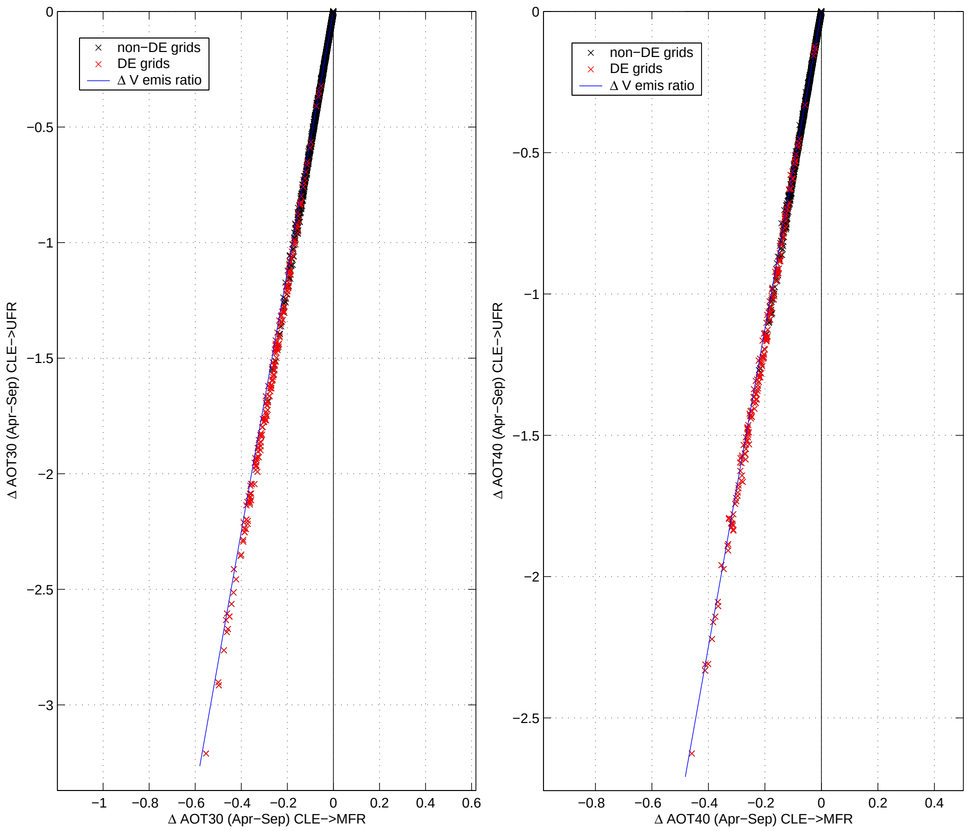

Figure 7.6: Changes in six-months AOT30 (left panel) and AOT40 (right panel) due to changes in German VOC emissions from CLE to MFR versus the changes from a reduction of German VOC emissions from CLE to UFR

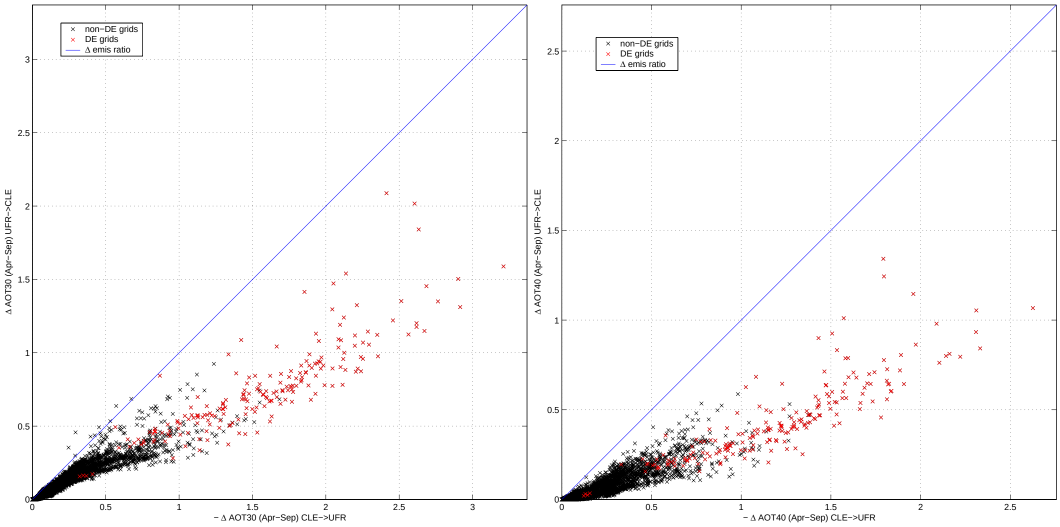

Figure 7.7: Differences in six months AOT30 (left panel) and AOT40 (right panel) resulting from a change of German NO_x, emissions from UFR to CLE with all other European emissions at UFR versus a change of the German NO_x emissions from CLE to UFR with all other European emissions at CLE

Because of the outstanding completion of the analysis of the 87 model experiments with the EMEP Eulerian model, as of now no final decision has been taken on the choice of AOT30 or AOT40 for implementation in RAINS. Table 7.4 summarizes the statistical performance of the two approaches related to monitoring data; differences in the variability of model results for further emission reductions will be further explored, and the importance of non-linearities (i.e., increasing ozone for NO_x reductions) will be evaluated for the AOT30 and AOT40, before a final decision is taken.

Table 7.4: Summary of the statistical performance of AOT30 and AOT40

	AOT30	AOT40
r^2 of critical level estimates		
for birch, beech *)	0.61	0.62
for oak	0.61	0.65
for Norway spruce and Scots pine	0.76	0.79
Correlation coefficient of ensemble dispersion models	0.65	0.61
Correlation coefficient of the EMEP model	0.57	0.48

*) used for definition of the proposed critical level for forest trees.

7.4 Atmospheric source-receptor relationships for AOT30 and AOT40

7.4.1 Linking critical levels with output of atmospheric dispersion models

Irrespective of the actual choice between AOT30 and AOT40, an integrated assessment needs to ascertain full consistency between the definition of the critical level and the output of atmospheric dispersion calculations. Critical issues are the height at which critical levels are defined and for which ozone is calculated, and the time intervals considered in the calculations.

The definition of the concentration-based critical level in the mapping manual applies to ozone concentrations at the top of the canopy, i.e., the upper boundary of the quasi-laminar layer (see UN/ECE, 2004). While the validation of dispersion model results is performed against monitoring data (typically between two and five meters height), ozone concentrations to be compared with critical levels for trees refer to typically 20 m height. The difference in ozone between measurement height and canopy height is a function of several factors, including wind speed and other meteorological factors, canopy height and the total flux of ozone. The mapping manual specifies appropriate approaches to apply vertical profiles for ozone, which are, *inter alia*, based on the EMEP deposition module (Emberson et al., 2000). To apply the correct values, model output from the EMEP model for a height of 20 meters is used.

It is important that the cumulative period over which the AOTX or $AF_{st}Y$ value is calculated is consistent with the period when the relevant crop, forest or semi-natural vegetation is actively growing and absorbing ozone. Thus, receptor specific time periods are defined in the mapping manual. In the absence of specific information, the value for six months with the highest value of AOTX should be used to indicate the maximum risk of exceedance for a particular receptor. The while the mapping manual provides a default table with country allocations to five different climatic zones (Table 7.5), for forest trees the default period of six months from April to September is suggested. Thus, RAINS will evaluate the AOTx excess for these six months for all countries in Europe.

The definition of the AOT refers to hourly mean values of ozone accumulated for all daylight hours, defined as hours with a mean clear sky global radiation above 50 Wm^{-2}. Further analysis will be necessary to identify the importance of latitudinal differences in daylight hours with high ozone concentrations to decide whether a uniform approach would be acceptable throughout Europe.

Table 7.5: Regional classification of countries for default time periods according to the Mapping Manual (UN/ECE, 2004)

Region	Possible default countries
Eastern Mediterranean	Albania, Bosnia and Herzogovina, Bulgaria, Croatia, Cyprus, Greece, FYR Macedonia, Malta, Slovenia, Turkey, Yugoslavia
Western Mediterranean	Italy, Portugal, Spain
Continental central Europe	Armenia, Austria, Azerbaijan, Belarus, Czech Republic, France[1], Georgia, Germany, Hungary, Kazakhstan, Krygyzstan, Liechtenstein, Moldova, Poland, Romania, Russian Federation, Slovakia, Switzerland, Ukraine
Atlantic central Europe	Belgium, Ireland, Luxembourg, Netherlands, United Kingdom
Northern Europe	Denmark, Estonia, Faero Islands, Finland, Iceland, Latvia, Lithuania, Norway, Sweden

[1] as an average between Western Mediterranean and Atlantic Central Europe

7.4.2 Source-receptor relationships for RAINS

In principle, the analysis for regional-scale source-receptor relationships will follow the approach on ozone modelling as described in the chapter on ozone health impacts. Differences emerge due to different ozone metrics and different reference heights. For health, the inlet height of monitoring stations (2-5 m) will be used, while for forests the definition of the critical level refers to concentrations at 20 m height. Obviously, for forests only regional scale calculations will be applied, which are considered representative for rural background locations, and no sub-grid modelling is foreseen.

Special emphasis will be given to the inter-annual variability of ozone formation.

7.5 References

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8 Modelling of acidification and eutrophication

8.1 The earlier RAINS approach

Back in 1999, RAINS used the concept of critical loads as a quantitative indicator for sustainable levels of sulphur and nitrogen deposition. This practical way to quantify a theoretical policy target for sustainability proved to be a powerful policy driver in the negotiations on the NEC Directive and the Gothenburg Protocol. Despite all theoretical shortcomings, the critical load concept was persuasive enough to justify differentiated emission control efforts and economic burdens across Europe.

The RAINS analysis using critical loads is based on the critical loads databases compiled by the Coordination Center on Effects under the UN/ECE Working Group on Effects. This database combines quality-controlled critical loads estimates of the national focal centres. As of 1999, this database contained details about 1,322,662 ecosystems (Posch et al. 1999). National focal centres have selected a variety of ecosystem types as receptors for calculating and mapping critical loads. For most ecosystem types (e.g., forests), critical loads are calculated for both acidity and eutrophication. Other receptor types, such as streams and lakes, have only critical loads for acidity, on the assumption that eutrophication does not occur in these ecosystems. For some receptors, like most semi-natural vegetation, only critical loads for nutrient nitrogen are computed, since the sensitivity to acidifying effects is less than the eutrophication effects.

The Coordination Center for Effects conducts quality control of the national estimates and provides, for each grid cell of the EMEP model, the cumulative frequency distribution of the critical loads for all ecosystems in the grid cell. Critical loads are provided as critical load functions, describing isolines of pairs of sulphur/nitrogen deposition that result in equal protection of ecosystems area in each grid cell (Posch et al. 1999, Hettelingh et al. 2001).

In the former RAINS model, deposition of nitrogen (N) and sulphur (S) is given as single values on the ($150*150km^2$) EMEP grid. Within a single EMEP grid cell, however, many (up to 100,000 in some cases) critical loads (CLs) for various ecosystems, mostly forest soils and surface waters, have been calculated. These CLs are sorted according to magnitude, taking into account the area of the ecosystem they represent, and the so-called cumulative distribution function (CDF) is constructed. This CDF is then compared to the single deposition values for that grid cell.

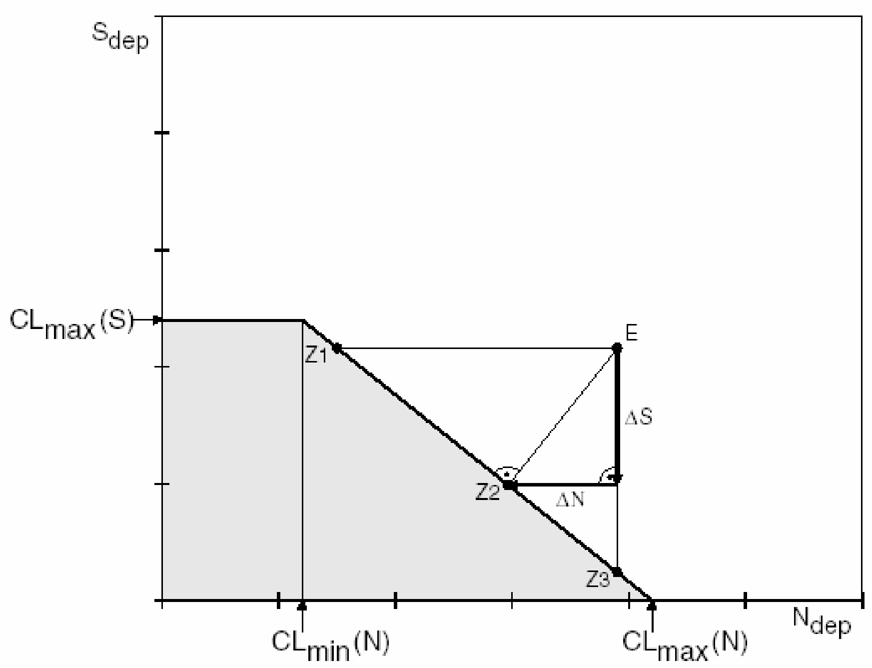

Figure 8.1: Example of a critical load function for S and acidifying N defined by the quantities CLmax(S), CLmin(N) and CLmax(N). It shows that no unique exceedance can be defined: Let the point E denote the current deposition of N and S. Reducing Ndep substantially one reaches point Z1 and thus non-exceedance without reducing Sdep; on the other hand one can reach non-exceedance by reducing Sdep only (by a smaller amount) until reaching Z3. For the purpose of the protocol negotiations, an exceedance has been defined as the sum of Ndep and Sdep (Δ S, Δ N), which are needed to reach the critical load function on the shortest path (point Z2).

In the integrated assessment for the 1994 Sulphur Protocol only sulphur was considered as acidifying pollutant (N deposition was fixed; it determined, together with N uptake and immobilization, the so-called sulphur factor). Furthermore, taking into account the uncertainties in the CL calculations, it was decided to use the 5-th percentile of the critical load CDF in a grid cell as the (only!) value representing the ecosystem sensitivity of that cell. And the difference between the (current) S-deposition and that 5-th percentile CL was called the exceedance of the critical load in that grid cell. This is illustrated in Figure 8.2a: Critical loads and depositions are plotted along the horizontal axis and the (relative) ecosystem area along the vertical axis. The thick solid and the thick broken lines are two examples of critical load CDFs (which have the same 5-th percentile critical load, indicated by ‘CL’). ‘D0’ indicates the (present) deposition, which is higher than the CLs for 85% of the ecosystem area. The difference between ‘D0’ and ‘CL’ is the critical load exceedance in that grid cell. Since it was impossible to reduce depositions in all European grid cells to critical loads (i.e. to reach zero exceedance), it was decided to reduce the exceedance everywhere by a fixed percentage, i.e. to "close the gap" between (present) deposition and (5-th percentile) critical load.

In Figure 8.2a, a deposition gap closure of 60% is shown as an example. As can be seen, a fixed deposition gap closure can result in very different improvements in ecosystem protection percentages (55% vs. 22%), depending on the shape of the CDF. In order to take into account the complete CDF of the critical loads (and not only the 5-th percentile), it was suggested to use an ecosystem area gap closure instead of the deposition gap closure. This is illustrated in Figure 8.2b: For a given deposition

‘D0’ to a grid cell the ecosystem area unprotected, i.e. with deposition exceeding the critical loads, can be read from the vertical axis. After agreeing to a certain (percent) reduction of the unprotected area (e.g. 60%), it is easy to compute for a given CDF the required deposition reduction (see ‘D1’ and ‘D2’ in Figure 8.2b). Another important reason to use the ecosystem area gap closure is that it can be easily generalized to two (or more) pollutants, which is not the case for the exceedance.

This generalization became necessary in the preparation for a new multi-pollutant/multi-effects protocol in the case of acidity critical loads, since both N and S are contributing to acidification. Critical load values are replaced by critical load functions (Figure 8.1) and percentiles are replaced by ecosystem protection isolines. The use of the area gap closure becomes problematic, however, if only a few critical load values or functions are given for a grid cell. In such a case, the CDF becomes discontinuous and (small) changes in deposition may result in either no increase in the protected area or large jumps in the area protected.

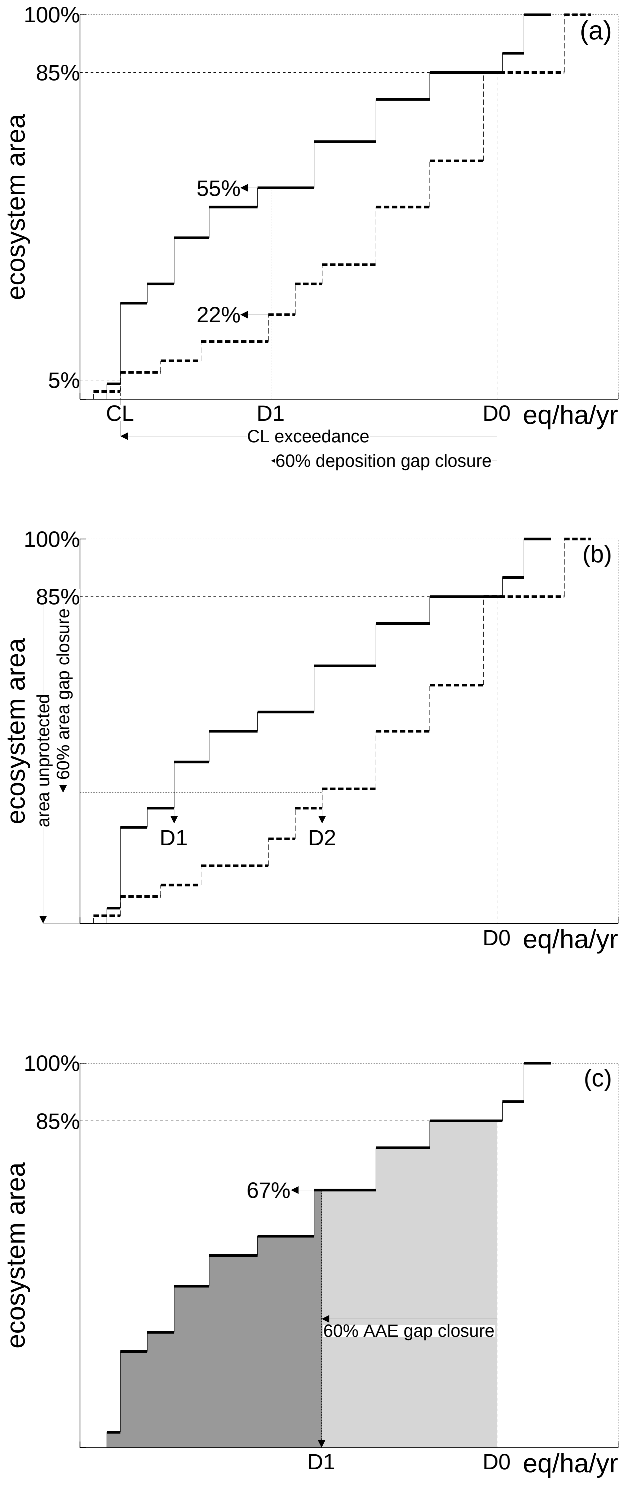

Figure 8.2: Cumulative distribution function (CDF; solid thick line) of critical loads (CLs) and the different methods of gap closure: (a) deposition gap closure, (b) ecosystem gap closure, and (c) accumulated exceedance (AE) gap closure. The dashed thick line in (a) and (b) depict another CDF, illustrating how very different ecosystem protection may result from the same deposition gap closure (a), or how different deposition reductions are required to achieve the same protection level. (Source: Posch *et al*., 2001).

To remedy the problem with the area gap closure caused by discontinuous CDFs, a new measure, the so-called accumulated exceedance (AE) has been introduced. This required the definition of an exceedance in the case of two pollutants: for a given deposition of N and S the exceedance is defined as the sum of the N and S deposition reductions required to achieve non-exceedance by taking the shortest path to the critical load function (see Figure 8.1). This exceedance is multiplied by the ecosystem area, and they are summed to yield the accumulated exceedance for a grid cell. In the case of one pollutant the AE is simply given as the area under the CDF of the critical loads (see grey-shaded area in Figure 8.2c). In addition, the average accumulated exceedance (AAE) has been defined by dividing the AE by the total ecosystem area of the grid cell, which has thus the dimension of a deposition. Deposition reductions are now negotiated in terms of an AE (or AAE) gap closure, which is illustrated in Figure 8.2c: a 60% AE gap closure is achieved by a deposition 'D1' which reduces the total grey area by 60%, resulting in the dark grey area; also the corresponding protection percentage (67%) can be easily derived. The greatest advantage of the AE is that it varies smoothly when depositions are varied, even for highly discontinuous CDFs, thus facilitating optimisation calculations in IAM. The advantages and disadvantages (shortcomings) of the three gap closure methods described above are summarized in Table 8.1.

Table 8.1: Advantages and disadvantages of the various approaches for critical loads

	Advantages	Disadvantages/Shortcomings
Deposition gap closure (used for 1994 UN/ECE Sulphur Protocol)	• Easy to use even for discontinuous CDFs (e.g. grid cells with only one CL)	• Takes only one CL value (e.g. 5-th percentile) into account • May result in no increase in protected area • Difficult to define for two pollutants
Ecosystem area gap closure (used for the EU Acidification Strategy)	• In line with the goals of CL use (maximum ecosystem protection) • Easy to apply to any number of pollutants	• Difficult (or even impossible) to define a gap closure for discontinuous CDFs (e.g. grid cells with only one CL)
Accumulated exceedance (AE) gap closure (used for the UN/ECE multi-pollutant multi-effects protocol)	• AE (and AAE) is a smooth and convex function of depositions even for discontinuous CDFs	• AE stretches the limits of the critical load definition (linear damage function!) • Exceedance definition not unique for two or more pollutants

Based on this information, RAINS used originally for the negotiations of the Second Sulphur Protocol the excess deposition of sulphur for the five percentile critical load ecosystem as a measure of the distance to the ultimate policy target (Tuinstra *et al.*, 1999). At that time the five-percentile ecosystem was chosen as the reference to safeguard the robustness of the estimates, so that extreme outliers, e.g., due to numerical artefacts caused by the limited resolution of land use maps, could not unduly influence the policy result. The 1994 Oslo Protocol established the 60 percent reduction of this excess deposition (the "60 percent gap closure") as its environmental target.

For the acidification strategy of the European Union in 1997 (European Parliament, 1998), RAINS adopted the multi-pollutant/multi-effect approach with the aim of finding an optimal balance of sulphur versus nitrogen reductions. Therefore, the sulphur-only approach of excess deposition for a given ecosystem was abandoned and the spatial dimension (ecosystems area) was introduced as the quantitative measure for the distance towards sustainability. In this way, the door was open for the sulphur/nitrogen critical load functions that allowed economic optimisation of emission controls for these two pollutants. In its acidification strategy, the Commission of the European Union called for a 50 percent reduction ("gap closure") of the area of ecosystems with acid deposition above their critical loads.

The subsequent policy deliberations raised concern about the robustness of this ecosystem area related metric, especially if ecosystems within a grid show only little variation in environmental sensitivity. In interactive discussions between the RAINS modellers and the representatives of the Member States in the EU Air Quality Steering Group, the concept of "accumulated excess deposition" was developed with the aim of maximising the robustness of the measure by integrating excess deposition over all critical loads estimates within a grid cell. For the Emission Ceilings Directive, RAINS used a 95 percent "gap closure" target for accumulated excess deposition of acidity, which turned out to be comparable to the ambition level of the 50 percent area gap closure target of the acidification strategy, but eliminated distortions due to some artefacts in critical loads estimates (Amann *et al.*, 1999).

8.2 New developments

As of now, there are no firm plans for a fundamentally different approach for using critical loads data for RAINS. The mapping community has a clear timetable to compile the latest estimates for the next round of analysis. If, with the forthcoming constellations of quantitative critical loads estimates and the realistic range for further emission reductions, the accumulated excess concept would not turn out to be useful, experience suggests that alternative concepts could be developed in close interaction with the decision makers and the effects community.

There are, however, a number of technical improvements (harmonization of land-use maps, eco-specific deposition) in the critical load mapping that call for slight modifications in the data handling.

Furthermore, dynamic acidification modelling has matured over the last years and it is a legitimate question to what extent this could and should be introduced into the integrated assessment modelling.

8.2.1 Harmonized land-use maps

In terms of technical improvements, land-use related issues were identified as major sources of uncertainties in the traditional implementation of critical loads modelling. First, the full consistency in land use data applied for critical loads estimates and for the atmospheric modelling of deposition was recognized to have prime influence on the accuracy of the integrated assessment. In the past, critical loads estimates produced by national experts have been derived from national land use maps with inconsistent classifications of land use types, while the EMEP model applied the land use map of the

Stockholm Environment Institute. The Coordination Center for Effects has pointed out major inconsistencies and started a process to harmonize land use maps used for the integrated assessment, based to the maximum possible extent on the CORINE inventory. IIASA will host a meeting of both communities to reach a practical and fast solution (March 10, 2004).

8.2.2 Ecosystem-specific deposition

In the past, scientific understanding did not allow the modelling of deposition to different ecosystems within a single grid cell on a mass-consistent basis. As a consequence, only the calculated grid-average deposition could be used to compare with site-specific critical loads. There is, however, ample evidence that, due to different surface roughness, deposition over forests is substantially higher than over open land, and thus systematically higher than the grid-average deposition.

With the new Eulerian model, EMEP has improved its deposition mechanism (TFMM, 2003) and can now provide ecosystem-specific deposition data. For the integrated assessment it has been agreed with MSC-W and the Coordination Center for Effects that critical loads data will be separated into different ecosystems types (forest, lakes, other vegetation), so that excess deposition can be more accurately calculated for specific ecosystems.

A preliminary assessment conducted by the Coordination Center for Effects based on the old critical loads data and the recent EMEP calculations suggests significantly lower levels of ecosystems protection especially for forest ecosystems. For acidification, the Gothenburg Protocol envisaged four percent to remain with acid deposition above critical loads. According to the new calculations, based on grid-average deposition, 15 percent of the ecosystems in the EU-25 would experience acid deposition above their critical loads, and 25 percent of the forests in the EU-15. For eutrophication the share of unprotected ecosystems would increase from 60 percent to 80 percent.

This major change in model estimates might pose fundamental questions about the robustness of quantitative scientific findings produced through integrated assessment. It has to be mentioned that all uncertainty and sensitivity assessments of the RAINS model have identified the use of grid-average deposition as one of the largest uncertainties in their evaluation, which introduced a systematic bias towards underestimating required emission reductions into the analysis. This finding was prominently communicated to decision makers (e.g., Amann *et al.*, 1999; Suutari et al., 2001) but was, in the absence of scientific ability to provide better results, accepted by the decision makers and taken into consideration during the negotiation phase.

8.2.3 Dynamic acidification modelling

Over the last years substantial progress has been made in the field of dynamic modelling of forest soils and freshwater bodies. A joint expert group on dynamic modelling has formed and has met four times to coordinate their activities and provide policy-relevant input for an integrated assessment. The group agreed that model testing has confirmed that given the same input data, all four models considered suitable for use in the forthcoming CCE call for data (MAGIC, SAFE, SMART, VSD) give similar outputs (Joint expert group on dynamic modelling, 2003). For soils, chemical recovery times can be estimated, while further work is necessary to model biological recovery. In waters, understanding of biological responses is sufficiently advanced that the lag time for organisms to recover after the chemical criterion is reached can be estimated. The biological recovery is affected by the rate of chemical recovery, by the generation time of the organisms and by stochastic processes.

For the acidification of soils and lakes, dynamic modelling allows, in principle, to extend the acidification analysis in RAINS beyond the critical loads approach. For critical loads, which reflect the steady-state situation of the dynamic acidification process, only two cases can be distinguished when comparing them to deposition:

- the deposition is at or below critical loads, i.e., does not exceed critical loads, and
- deposition is greater than critical loads, i.e., there is a critical load exceedance.

In the first case, there is no apparent risk of ecosystems damage, i.e., no reduction in deposition is deemed necessary. In the second case there is, by definition, an increased risk of damage to the ecosystem. Thus, a critical load serves as a warning as long as there is exceedance. However, it is often assumed that reducing deposition to (or below) critical loads immediately removes the risk of harmful effects, i.e., that the chemical criterion that links the critical load to (biological) effects immediately attains a non-critical (safe) value and that there is immediate biological recovery as well.

Dynamic models estimate the time required to attain a certain chemical state in response to deposition scenarios. In addition to the delay in chemical recovery, there is likely to be a further delay before the original biological state is reached. Five stages (or phases) can be defined in the temporal acidification and recovery process (Posch et al., 2003):

- Deposition was, and is, below the critical load (CL) and the chemical and biological variables do not violate their respective criteria (the ideal situation).
- Deposition is above CL, but chemical and/or biological criteria are not violated because there is a time delay before this happens. No damage is likely to occur at this stage, despite exceedance of the CL.
- Deposition is above CL, and both the chemical and biological criteria are violated. Damage occurs.
- Due to emission reductions, deposition is again below CL, but the chemical and biological criteria are still violated and thus recovery has not yet occurred.
- Deposition is below CL, and both criteria are no longer violated. Only at this stage can the ecosystem be considered to have recovered.

In this system, the *damage delay time* (length of phase 2) and *recover delay time* (length of phase 4) are important variables, which provide relevant information for emission control strategies.

The most straightforward use of dynamic models for an integrated assessment is for scenario analysis: the future chemical (and biological) status of an ecosystem is evaluated for a prescribed future deposition pattern. This is very simple for selected sites and requires only minor extra effort for a large number of sites. The results of a scenario analysis can then guide stakeholders in their quest for further deposition reductions. This relatively slow process could be accelerated and rationalized with the optimisation approach, in which the environmental targets are determined with dynamic models. For this purpose, dynamic models need to be linked with the integrated assessment model either through full integration of the dynamic models into the IAM or through dynamic model output (response functions) that can be used in optimisation. An interface between dynamic models and RAINS in the form of “target load functions” has been developed in cooperation with the Coordination Center for Effects (Posch et al., 2003). These target load functions provide isolines of pairs of sulphur/nitrogen deposition for a given target year that achieve recovery of a given ecosystem within a given time interval. Such functions have also been developed for multiple ecosystems within a grid cell in cooperation with the Joint Expert Group (JEG) on Dynamic Modelling (Posch et al. 2003). For optimisation, such functions allow the derivation of target deposition levels that would lead to chemical recovery of *x* percent of the ecosystems within *y* years. Once a target year for emission reductions has been decided, the variables *x* and *y* are then subject to policy choice. Due to the lack of actual output from dynamic models, this interface has not yet been applied in practice for RAINS calculations.

While noting the general progress in dynamic modelling and accepting the need for further scientific insight into important mechanisms (e.g., the role of nitrogen in ecosystems), some issues relevant for the use of results in integrated assessment modelling remain to be clarified:

- Dynamic modelling has been mastered for individual soil or lake ecosystems. Up-scaling from single sites to regions/grid squares is in itself a major task, especially if dynamic models are not implemented for all ecosystems in a grid cell.
- Collection of further data might be necessary to apply this target load function approach for all European ecosystems.
- An important strength of critical load data in past applications was their complete coverage of all ecosystems in Europe, which allowed the policy analysis to be free of observational bias due to missing information. Especially the uniform gap closure concept that was applied to all European ecosystems turned out as a strong policy argument: the perceived equal (relative) environmental improvements justified inequities in economic efforts to reduce emissions. If results from dynamic modelling will not be available for all ecosystems, concerns about the objectivity of the choice of the selected sites might become an obstacle to using dynamic modelling results as immediate targets for international environmental policy.
- While IIASA has led a study on historic deposition of sulphur and nitrogen from 1880 to 2030 (Schöpp *et al.*, 2003), which provides essential input to the dynamic models, corresponding information on historic base cation deposition is missing, which introduces a potentially major source of uncertainties into model calculations.
- By their nature, dynamic models cover periods of several decades up to 100 years. To simulate future recovery processes, the impact of climate change should not be ignored.

Given these unresolved issues, it is planned to use results from dynamic modelling for the "scenario analysis" mode in RAINS, i.e., to illustrate the consequences of otherwise determined emission reductions on the recovery of forest soils and lakes. Use of the results from dynamic modelling for a limited number of sites as policy targets in the optimisation seems premature, given the unresolved issues listed above and the potential implications on the robustness of model results. However, it seems perfectly feasible to continue defining targets for the RAINS optimisation on the basis of critical loads, to evaluate the optimal set of emission reductions along their recovery times for soils and lakes and, if the resulting recovery times turn out to be politically unacceptable, to tighten the targets in relation to the critical loads.

8.3 Atmospheric modelling of acid deposition

The recent review of the new EMEP Eulerian model concluded that

- *"There was high confidence in the EMEP model's ability to represent the broad spatial patterns in the deposition of sulphur and oxidized nitrogen compounds across Europe;*
- *While a spatial resolution of 50 km x 50 km represents a major improvement compared with the EMEP Lagrangian model, considerable sub-grid scale variations can still be expected and so some additional statistical treatment will be required to account for in-square variations;*
- *There was every confidence in the model's ability to reproduce the observed trends in sulphur and oxidized nitrogen deposition;*
- *There was limited confidence in the model's ability to represent the spatial pattern and trends in reduced nitrogen deposition because of the lack of understanding of the fate and behaviour of ammonia and the difficulties associated with the model representation of ammonia emissions and deposition."*

To explore the response of the recent version of the EMEP Eulerian model towards changes in precursor emissions, the same 87 model experiments with the EMEP Eulerian model as described in the PM chapter have been performed and the responses of various deposition metrics have been investigated. Of particular interest was the detection of potential non-linearities that would preclude the use of simple linear source-receptor matrices for the calculations in RAINS. While the new EMEP model provides dry deposition estimates for a range of different land-use classes, lack of the underlying land-use information did not allow this first analysis to explore deposition other than grid average. Once this information will be obtained from EMEP, the analysis presented below will be repeated for deposition to deciduous and coniferous forests.

The following graphs compare changes of calculated annual deposition of the various acidifying compounds for the UK grid cells (red crosses) and other European receptors (black crosses) resulting from changes in UK emissions.

Figure 8.3 (left panel) shows the ratio between changes in SO_2 emissions and resulting changes in sulphur deposition for (1) change from CLE to MFR, (2) change from CLE to UFR. There is an almost perfect linear relationship, both for receptors close to the sources and remote sites. As demonstrated in the right panel, the response is almost independent of the overall pollution level. Although not shown here, there is a very small impact on sulphur deposition if NO_x emissions are reduced, and a more noticeable effect when NH_3 is reduced, which warrants further investigation. When emissions of all pollutants are modified simultaneously, the sulphur deposition response appears to be independent of the overall pollution level (Figure 8.4). On this basis, the use of linear source-receptor relationships seems appropriate to reproduce the response in sulphur deposition calculated with the full EMEP Eulerian model.

For the deposition of oxidised nitrogen compounds, similar findings emerge. There is rather good linearity for changes in NO_x emissions (Figure 8.5). These are virtually independent of changes in SO_2 emissions, but show a noticeable dependency towards isolated changes in NH_3 (Figure 8.6). Again, if the various pollutants are reduced in an ensemble, a linear description seems to perform very well (Figure 8.7).

The response of deposition of reduced nitrogen towards changes in NH_3 emissions is extremely linear (Figure 8.8). Single-pollutant changes of NO_x emissions exert a very small disturbance, which is however below 0.5 percent and disappears if emissions are reduced in an ensemble (Figure 8.9).

From this preliminary analysis a representation of the source-receptor relationships for acid deposition resulting from the full Eulerian EMEP model through linear functions would seem to be an acceptable approach for integrated assessment. The caveat applies that this finding needs to be confirmed for forest-specific deposition.

The Mapping Manual of the ICP on Modelling & Mapping" also requests that calculations on the excess of critical loads need to be based on multi-year meteorology, in order to exclude the influence of inter-annual variability. For the earlier policy application of RAINS, source-receptor relationships were computed for 10 meteorological years, and the average relationships were used for calculations in RAINS. This inter-annual variability is indeed an important factor and needs to be considered in an integrated assessment. Thus, it is the plan to use calculations for as many meteorological years as possible for the analysis. The practical availability of EMEP model results will determine what can be done for RAINS.

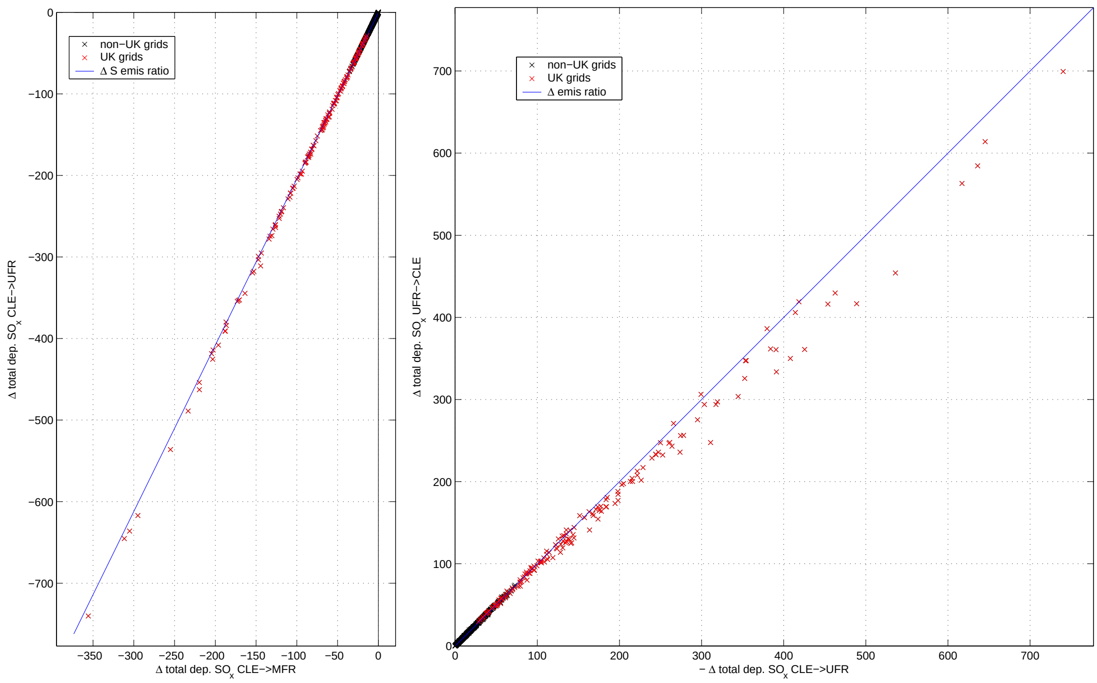

Figure 8.3: Left panel: Change of total sulphur deposition (dry + wet) due to changes in the UK SO_2 emissions from CLE to MFR versus the deposition changes resulting from a reduction of UK SO_2 emissions from CLE to UFR. Right panel: Differences in total sulphur deposition (dry + wet) due to changes in the UK SO_2 emissions from UFR to CLE with all other European emissions at UFR, versus a change of the UK SO_2 emissions from CLE to UFR with all other European emissions at CLE.

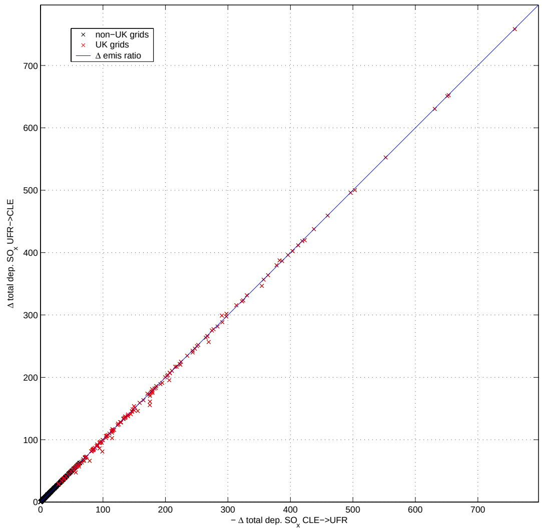

Figure 8.4: Differences in total sulphur deposition (dry + wet) due to changes in the UK SO_2, NO_x and NH_3 emissions from UFR to CLE with all other European emissions at UFR, versus a change of the UK emissions from CLE to UFR with all other European emissions at CLE.

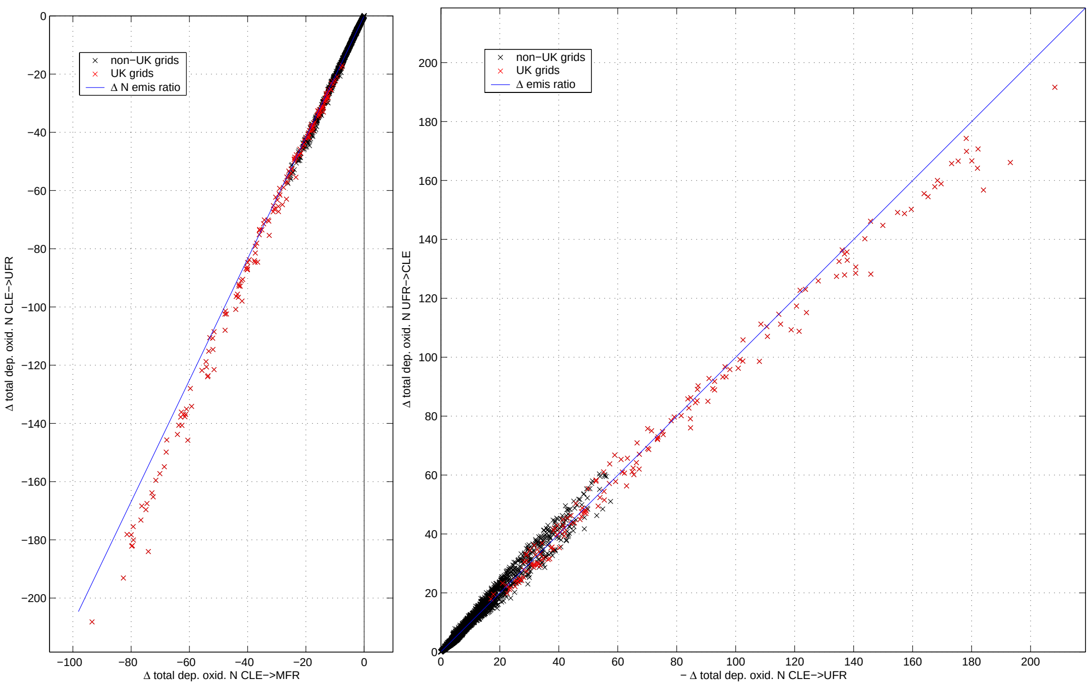

Figure 8.5: Left panel: Change of deposition of oxidised nitrogen (dry + wet) due to changes in the UK NO_x emissions from CLE to MFR versus the deposition changes resulting from a reduction of UK NO_x emissions from CLE to UFR. Right panel: Differences in total deposition of oxidised nitrogen (dry + wet) due to changes in the UK NO_x emissions from UFR to CLE with all other European emissions at UFR, versus a change of the UK NO_x emissions from CLE to UFR with all other European emissions at CLE.

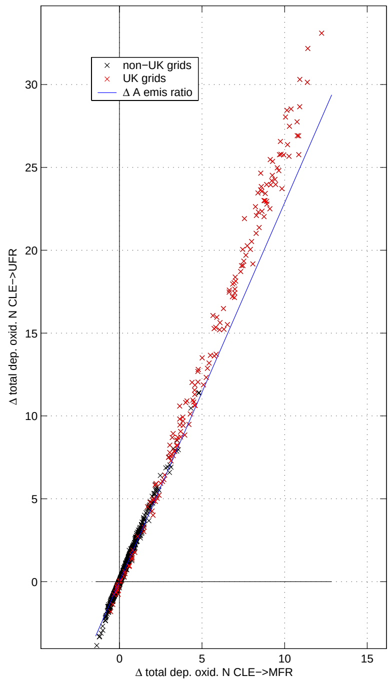

Figure 8.6: Change of deposition of oxidised nitrogen (dry + wet) due to changes in the UK NH_3 emissions from CLE to MFR versus the deposition changes resulting from a reduction of UK NH_3 emissions from CLE to UFR

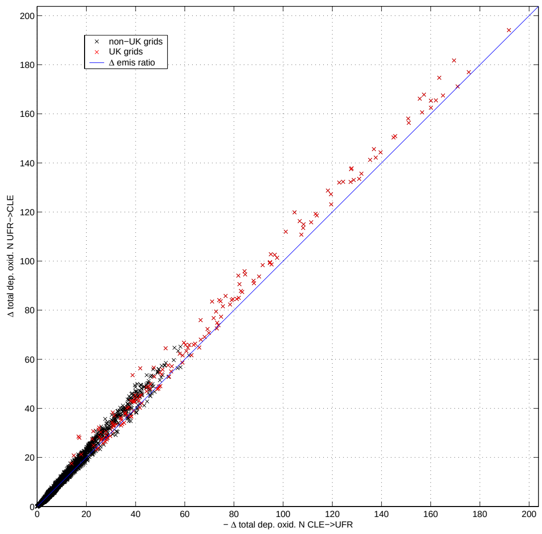

Figure 8.7: Differences in deposition of oxidised nitrogen (dry + wet) due to changes in the UK SO2, NOx and NH3 emissions from UFR to CLE with all other European emissions at UFR, versus a change of the UK emissions from CLE to UFR with all other European emissions at CLE.

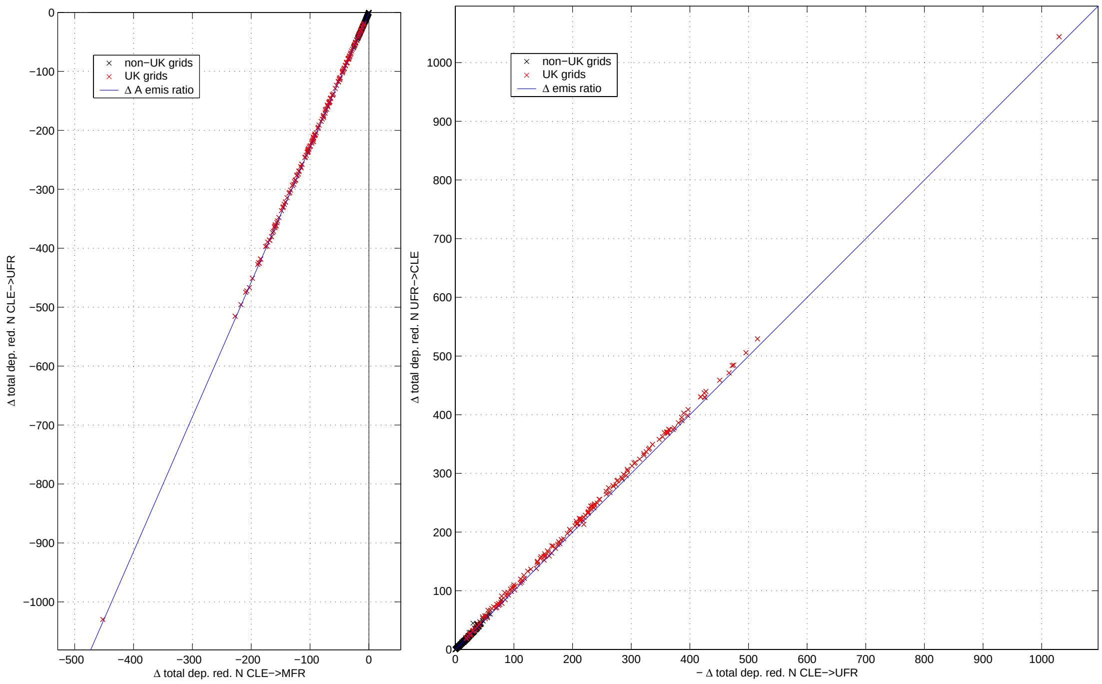

Figure 8.8: Left panel: Change of deposition of reduced nitrogen (dry + wet) due to changes in the UK NH_3 emissions from CLE to MFR versus the deposition changes resulting from a reduction of UK NH_3 emissions from CLE to UFR. Right panel: Differences in total deposition of reduced nitrogen (dry + wet) due to changes in the UK NH_3 emissions from UFR to CLE with all other European emissions at UFR, versus a change of the UK NH_3 emissions from CLE to UFR with all other European emissions at CLE.

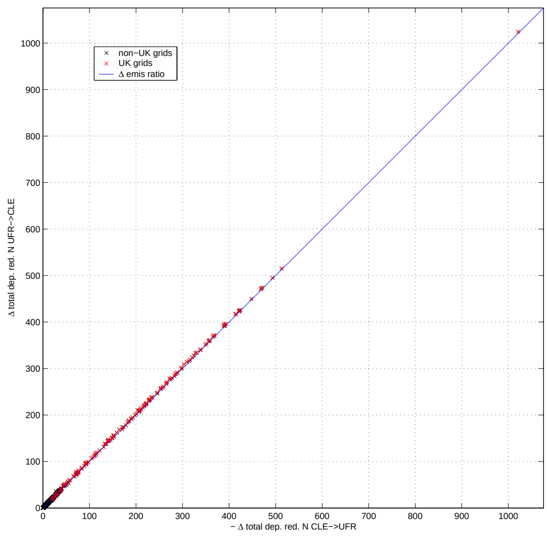

Figure 8.9: Differences in deposition of reduced nitrogen (dry + wet) due to changes in the UK SO_2, NO_x and NH_3 emissions from UFR to CLE with all other European emissions at UFR, versus a change of the UK emissions from CLE to UFR with all other European emissions at CLE.

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9 Uncertainties

9.1 Introduction

The adequate treatment of uncertainties is a persistent and critical issue in integrated assessment modelling. Recent years saw a lively debate in the scientific literature about numerous theoretical aspects of uncertainties and integrated assessment (Rotmans and van Asselt, 2001), especially in the context of climate change. Uncertainties in integrated assessment models were also of practical concern when model results were used in international environmental negotiations, e.g., on transboundary air pollution in Europe.

In contrast to the vivid interest from the academic community and the users of results of integrated assessments, there are only few publications where developers of integrated assessment models systematically analyse uncertainties in their models and develop practical approaches for treating uncertainties in an adequate manner. There are a number of reasons for this shortcoming. With integrated assessment models, a comprehensive uncertainty analysis is not a trivial issue, and it takes considerable intellectual and computational resources to develop and practically apply appropriate techniques. In a typical resource-constrained situation, integrated assessment modellers often expect larger insights from integrating additional aspects into their model or improving existing data and models than from a cumbersome uncertainty assessment. Secondly, there is no ready-made methodology available that could be directly used for the analysis of uncertainties in integrated assessment models. As Rotmans and van Asselt, 2001, observe, "*Uncertainty analysis lacks a tool kit to address salient uncertainties in an adequate manner as a central activity in integrated assessment modelling*". Third, it is also not obvious which type of information about uncertainties is meaningful in the policy context. Quantitative standard measures of uncertainties like confidence intervals might not really tell much, e.g., to users of IAMs, if they want to use results for practical policy decisions.

The Regional Air Pollution Information and Simulation (RAINS) model is a prominent example of an integrated assessment model that found practical application in a series of international environmental negotiations. *Inter alia*, RAINS was used in recent years to provide practical policy guidance to the negotiations on the 1999 Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone of the UN/ECE Convention on Long-range Transboundary Air Pollution, and the European Commission used RAINS to quantify its proposed emission ceilings for the Emission Ceilings Directive of the European Union. At that time, RAINS did not contain a module that explicitly addressed uncertainties in a quantitative way. Not surprisingly, negotiating Parties and stakeholders raised the uncertainties of model calculations as an issue of concern, be it from real concern or, in several instances, as an additional argument against action that was opposed by a Party or the industry for other reasons.

While a robust treatment of uncertainties was from the beginning not only in the scientific interest, but also in the pure self-interest of the model developers, special concerns about uncertainties raised by Parties during the negotiations were addressed through a series of modifications to the model, through adapted definitions of environmental targets and a series of sensitivity analyses. This paper reviews the various ways in which uncertainties were managed in the construction and use of the RAINS model for the Gothenburg Protocol and the Emission Ceilings Directive.

9.2 Uncertainties in the RAINS Model

Like all models, the RAINS model attempts to develop a holistic understanding of a complex reality through a variety of reductionistic steps. This simplification process is burdened with many uncertainties related to methodological issues, lack of understanding and insufficient data. Thus, there exist considerable uncertainties in almost all parts of the model framework, e.g., in the emission inventories, the estimates of emission control potentials, the atmospheric dispersion calculations and set-up of the model, with important input assumptions, and with the available data. Table 9.1 provides an incomplete list of uncertainties associated with the RAINS model.

Recognizing the potentially critical influence of such uncertainties, a number of uncertainty analyses addressing individual aspects of these uncertainties have been undertaken during the development and application of the RAINS model:

Sorensen, 1994a,b conducted a sensitivity analysis for the cost calculation routine implemented in the RAINS model and explored how such uncertainties affect the outcome of an optimisation analysis. In general, quantitative optimisation results were found to be sensitive to variations in the capacity utilization of boilers and in the sulphur contents of fuels. While such variations might change results for individual countries, overall optimised patterns of required emission reductions, however, do not change significantly.

Altman *et al.,* 1996 analysed the influence of uncertainties in emission control costs on calculations of cost-effective European sulphur emission reductions. A specialized solution procedure was developed and a number of different cost curves were generated to model the uncertain costs.

An analysis of the robustness of RAINS-type cost curves (Duerinck, 2000) suggested that uncertainties in the cost components, although relatively high, were much less important for the overall uncertainty than uncertainties in the emissions.

The relationship between deposition targets and the calculated emission ceilings for Denmark has been investigated using Monte Carlo simulation (Bak and Tybirk, 1998). In addition, the sensitivity of the calculated emission ceilings with respect to changes in Danish national data has been analysed. The analysis explored the sensitivity towards modifications in the energy scenario, the agricultural scenario, the ammonia emission factors and the marginal costs of SO_2, NO_x and ammonia abatement.

Alcamo *et al.*, 1987 explored to what extent interregional transport of air pollutants in Europe could be described by linear relations. It was found that the linearity between emissions and deposition strongly depends on the distance between emitter and receptor, the averaging period, the constituent (acidity, oxidants, sulphur, etc.), and the form of deposition (e.g., whether total deposition is considered or wet deposition alone).

The same authors addressed the uncertainty of atmospheric source-receptor relationships for sulphur within Europe (Alcamo and Bartnicki, 1990). Stochastic simulation was used to compute the effect on selected transfer coefficients of uncertainties related to transport wind, meteorological forcing functions, model parameters and the spatial distribution of emissions. Uncertainty estimates for 30 source-receptor combinations – based on one year's meteorological conditions – suggested a relative uncertainty of 10 percent to 30 percent in the transfer coefficients, not correlated with the distance between emission source and receptor. However, their absolute uncertainty (standard deviation) was found strongly correlated with distance and proportional to the values of the transfer coefficients themselves.

Table 9.1: Taxonomy of uncertainties in the RAINS model

Model structure	Emission calculations	Selected sectoral aggregation
		Determination of mean values
	Atmospheric dispersion	Linearity in atmospheric dispersion
		Selected spatial resolution, ignoring in-grid variability
		Country size (country-to-grid)
	Critical loads estimates	The threshold concept, e.g., the critical Ca/Al ratio
		Selected aggregation of ecosystems
		Static representation of a dynamic process
Parameters	Emission calculations	Expected values for fuel quality, removal efficiencies and application rates
	Atmospheric dispersion	Expected values of parameters for describing chemical and physical processes (conversion rates, deposition rates)
		Mean transfer coefficient in view of inter-annual meteorological variability
	Critical loads estimates	Expected values of base cation deposition and uptake, throughflow, nitrogen uptake in critical loads calculations
Forcing functions	Emission calculations	Accuracy of statistical information on economic activities
		Projections of sectoral economic activities
		Future implementation of emission controls
	Atmospheric dispersion	Spatial distribution of emissions within countries
		Accuracy of meteorological data
Initial state	Emission calculations	Uncontrolled emission factors
		State of emission controls in the base year
	Atmospheric dispersion	Natural emissions
		Hemispheric background

Hettelingh, 1989 addressed the uncertainty of modelling regional environmental impacts caused by imperfect compatibility of models and available measurement data. He concluded that an uncertainty analysis of integrated environmental models, which integrates different processes (e.g., meteorological, soil and watershed acidification processes) with a probabilistic interpretation of model predictions, might allow different models and data to provide overlapping confidence intervals.

The uncertainty in ecosystem protection levels in Finland was found to be dominated by the uncertainties in critical loads for most parts of the country (Syri *et al.*, 2000).

While these studies addressed uncertainties inherent in individual elements of the RAINS model, the question how these uncertainties interact in an integrated assessment model received less attention. Van Sluijs, 1996 compared different approaches to the management of uncertainties taken by regional integrated assessment models for climate change and regional air quality. A comprehensive treatment of uncertainties turned out to be a challenge for all models available at that time: (i) Models did not fully address all relevant aspects within the whole spectrum of types and sources of uncertainty; (ii) they failed to provide unambiguous comprehensive insight to both the modeller and the user into the quality and limitations of models and their answers and (iii) they failed to address the subjective component in the appraisal of uncertainties.

This finding did not come as a surprise to the developers of integrated assessment models, since it demonstrated that, due to the multi-dimensional complexity of such models, an appropriate treatment of uncertainties is far from trivial. The computational complexity of the RAINS model system made it difficult to conduct a formal uncertainty analysis with traditional approaches (e.g., a Monte-Carlo analysis) that would yield quantitative insight. For instance, because a single optimisation run of the non-linear ozone model consumed approximately four days of CPU time at the fastest computer available at IIASA, it was completely out of reach to conduct the large number of model runs required for such analysis within the time scale of the negotiations. A single set of Monte-Carlo runs would have taken several decades of CPU time to complete. A further dimension of the complexity was that only insufficient quantitative information about the uncertainties of the input data was available and modellers would have had to make bold assumptions about error distributions and the independence of parameters, which would, themselves, constitute further sources of uncertainty.

9.3 Use of RAINS in international negotiations

The need for an integrated assessment model to provide a scientific basis for emission reductions under the Convention on Long-range Transboundary Air Pollution was apparent by 1985. For various reasons, the Executive Body decided to have the RAINS model perform most of the analyses underlying the negotiations over the second sulphur protocol. The results of two other models, i.e., the Abatement Strategies Assessment Model (ASAM) developed by Imperial College, London, and the Coordinated Abatement Strategy Model (CASM) of the Stockholm Environment Institute, were used for comparisons. RAINS was more fully developed than the two other models at the start of the negotiations; its results would be credible to countries both in Eastern and Western Europe because IIASA is an international institute, and a 1991 workshop on the model’s usability had already given potential users some familiarity with it (Tuinstra *et al.*, 1999).

9.3.1 The negotiations on the Second Sulphur Protocol

To give the negotiations over the second sulphur protocol a firm scientific basis, the Executive Body formed a special task force on integrated assessment modelling to determine the optimal amount of emission reductions (and their distribution over space) as well as the costs and benefits associated with those reductions. The simulations themselves were conducted in close collaboration with the Working Group on Strategies, where the negotiations between countries took place. Typically, the working group would request a particular set of simulations, study the results, and then request different or more refined simulations. The emission reductions of the ultimate scenario, aiming at closing the gap between 1990 deposition and the critical loads by 60 percent, were accepted by most countries as their obligations laid down in the protocol in 1994.

A number of limitations inherent in the RAINS model were recognized during the negotiations as potential sources of uncertainties. These included (i) the usual uncertainties in the data, including gaps in observable data and the necessary imprecise predictions of future development, (ii) the steady-state nature of the critical loads concept, which ignores the dynamic nature of biological and chemical processes in nature, and (iii) the fact that RAINS did not include structural changes in the energy sector as a means for controlling emissions. At the beginning of the negotiations over the second sulphur protocol, there was a good deal of concern about the uncertainties in the simulations with RAINS and the other models. As time went on, however, the negotiators shifted their attention to the political assumptions and other constraints adopted in the optimisation exercises. One reason for this shift may have been the growing realization that the uncertainties in such inputs were greater than those in the models themselves. It is also possible that some delegates were simply raising concern about uncertainty as a means of delaying a vote on the protocol until there was more political support at home (Tuinstra *et al.*, 1999).

9.3.2 The Gothenburg protocol and NEC directive

After signature of the second sulphur protocol in 1994, the Convention on Long-range Transboundary Air Pollution focused on the revision of the first NO_x protocol signed in 1986, which entered into force in 1998. By that time scientific evidence had demonstrated that emissions of nitrogen oxides have multiple effects on the environment. Most notably, they contribute to acidification and eutrophication (excess fertilization) of terrestrial and aquatic ecosystems, and they play a central role in the formation of ground-level ozone. The fact that, under certain conditions, in urban ozone plumes reductions of NO_x emissions could lead to further increases in ozone, was recognized as a hurdle in designing NO_x reduction strategies.

The multi-pollutant/multi-effect concept of the RAINS model offered a clear concept for addressing the multi-facetted nature of NO_x controls: RAINS offered an operational method for designing emission control strategies that simultaneously addressed acidification, eutrophication and ground-level ozone. The model facilitated the development of coordinated emission controls that yielded benefits to all parties for all three environmental effects, despite the potentially counter-productive response of ozone formation to reductions in NO_x emissions.

A workshop organized by IIASA for the negotiators of the Convention on Long-range Transboundary Air Pollution clearly demonstrated the strong need for considering further controls of sulphur dioxide emissions, if such multi-pollutant/multi-effect strategies were to be cost-effective. Although after the recent signature of the second sulphur protocol SO_2 was not on the immediate agenda of the negotiations under the Convention, Parties accepted this requirement and agreed to strive for a multi-pollutant/multi-effect protocol including SO_2 emissions.

It is important to realize that the evidence provided by integrated assessment models about the potential gains of an integrated approach in a complex situation where misbalanced emission controls could lead to a deterioration of environmental conditions convinced decision makers to revise their political agenda and to use integrated assessment models for exploring cost-effective solutions. It is also clear that this decision was taken in full awareness – and acceptance - of the uncertainties of integrated assessment models.

While the Convention was preparing for a multi-pollutant/multi-effect protocol, the European Community, after the accession of Sweden, Finland and Austria, embarked on parallel discussions to orient its further clean air policy. At the council meeting in March 1995, Sweden requested a report from the EU Commission that would include an assessment of the impact of current and proposed EU legislation on acidification and, as a follow-up to that report, an acidification strategy of the EU. Until then, key personnel in the Commission and the Environment Directorate had almost exclusively favoured a "best available technology" approach (Wettestad, 2002). This approach, primarily developed by Germany, is more interested in the emissions side of the issue and its related technological options than the environmental side, as demonstrated in the thinking on critical loads. For its communication on a EU acidification strategy (CEC, 1997), the Commission used the RAINS integrated assessment model to build an analytical bridge between the techno-economic aspects of emission control strategies and their environmental impacts. The concept was appreciated by the European Parliament and the Council, and in 1997 the Commission began to prepare a proposal on a Directive on National Emission Ceilings using the RAINS model as the central analytical tool for deriving quantitative emission caps for the Member States.

Thus, from 1997 to 1999, analysts at IIASA used the RAINS model for a large number of iterative scenario analyses. This was done in close interaction with negotiators at the Working Group on Strategies of the Convention on Long-range Transboundary Air Pollution and, in parallel, with the staff at the European Commission and the various working groups established by the Commission involving representatives of Member States and other stakeholders. In total, the analysis ran through 11 iterations, where decision makers requested model calculations, analysed their results and suggested modified scenario runs. All reports produced by IIASA for these negotiations are freely available on the Internet (http://www.iiasa.ac.at/~rains). In addition, IIASA developed an on-line version of the RAINS model that allowed all stakeholders to explore their own scenarios in an interactive way. During the negotiations, more than 8000 scenarios were calculated over the Internet.

The shared use of IIASA's RAINS model for both activities helped to maintain consistency between the work of the Convention on Long-range Transboundary Air Pollution and the European Commission. Both bodies explored with similar assumptions and environmental objectives, although the analysis for the Convention obviously also included countries that are not members of the EU. Finally, both processes resulted in similar emission ceilings, with the emission ceilings directive of the EU being slightly more demanding than the Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone. The interplay of these two policy processes is analysed in Wettestad, 2002, also providing detailed comparisons on the obligations for individual countries.

9.3.3 Model uncertainties and negotiations

During the negotiations, iterations of scenarios for environmental interim objectives explored different ambition levels along their economic implications and assessed the robustness of model results in a variety of ways. In these discussions between stakeholders and the model developers, which took place at the Working Group on Strategies and the Task Force on Integrated Assessment Modelling of the Convention as well as with the Air Quality Steering Group established by the EU Commission, the proper treatment of uncertainties and their implications on policy conclusions were extensively discussed.

The European Parliament, when discussing the Communication of the Commission about the Acidification Strategy, was concerned about undue impacts of uncertain model elements on emission control requirements as calculated by the RAINS model. In particular, the use of hard environmental constraints for each specific grid-cell to drive Europe-wide emission reductions raised concerns about the reliability of the underlying critical loads estimates. Uncertainty was raised as a matter of concern by industry and countries when the RAINS model was used to guide negotiations under the Convention on Long-range Transboundary Air Pollution on the Gothenburg Protocol and the proposal of the Commission of the European Union on a Directive on National Emission Ceilings (e.g., Cocks *et al.*, 1998).

9.4 Control of uncertainty through model design

The developers of the RAINS model were aware from the beginning that, although uncertainties are an important aspect in the policy debate, a formal treatment of uncertainty using traditional methods would be difficult to conduct. Instead, the model developers decided to consider uncertainty management as an important guiding principle already during the model development phase and adopted a variety of measures in model design and scenario planning to systematically minimize the potential influence of uncertainties on policy-relevant model output (Schöpp *et al.*, 1999). For instance, at all phases of model development and use, explicit confidence intervals (for emission control potentials, deposition ranges, ozone levels, ecosystems sensitivities, etc.) defined the range within which the model was proven to work with sufficient accuracy. Potential reliance of optimised solutions on single point estimates were avoided through integral measures for environmental sensitivities (e.g., accumulated excess of critical loads, long-term ozone measures, etc.). Specially designed compensation mechanisms allowed controlled violation of environmental targets for single ecosystems with potentially uncertain sensitivities. Wherever possible, preference was given to relative model outcomes (comparing two model outputs) rather than to absolute values. For ground-level ozone, less weight was given to extreme meteorological situations because their representativeness was questionable and the performance of the meteorological model for such rare situations was less certain. Sensitivity analysis attempted to identify systematic biases and showed that with large probability the emission reductions resulting from the model calculations could be considered as minimum requirements, suggesting that there is only little chance that policy measures suggested by the model needed to be revised in the future in the light of new information.

9.4.1 Use of atmospheric dispersion models

In RAINS, atmospheric dispersion is modelled by means of an emission-deposition transfer matrix, calculated as a mean over several years (in practice, 11) in order to reduce the effect of temporal meteorological variability. Analysis of the meteorological variability (Suutari *et al.*, 2001) showed that the relative uncertainty in the mean transfer coefficients for sulphur and oxidized nitrogen deposition (for the emissions of 1990) is, in most parts of Europe, less than 10% at the 95% confidence level. The corresponding uncertainty, due to meteorological variability, in deposition resulting from NH_3 emissions is lower still, nowhere more than 5%.

Although the EMEP model, which provides the transfer coefficients used in RAINS, quantifies very low deposition resulting from rare modelled events, such small elements (<0.5 mg m^{-2} for 1990 emissions), which carry a high degree of uncertainty, are ignored in RAINS calculations.

RAINS uses a 'reduced-form' model (Heyes *et al.*, 1996) of the source-receptor relationships between the NO_x and VOC precursor emissions and ozone concentrations. These are derived from the EMEP photo-oxidants model (Simpson, 1993), which has participated in a number of model intercomparisons (Derwent, 1993, Andersson-Sköld and Simpson, 1997) and whose results have been compared with available ozone measurements (Simpson, 1993, Malik *et al.*, 1996). For all politically important scenarios, the results from RAINS were confirmed with the full EMEP model.

9.4.2 Quantitative indicators for environmental impacts

Within the integrated assessment process the long-term effects of ozone exposure have been assessed in terms of the cumulative AOT (accumulated over threshold) concept (Kärenlampi and Skärby, 1996). This integral measure minimizes the reliance of single point estimates. However, such a measure involves a cut-off concentration (40 ppb for vegetation damage, 60 ppb in relation to human health) below which ozone concentrations do not contribute to the AOT value. To allow for the uncertainty in modelled ozone concentrations, a probabilistic approach was taken, involving use of a sigma function to calculate expected values, in deriving AOT values from the EMEP model output concentrations.

The assessment of the environmental impact of acidification and eutrophication uses the concept of accumulated excess acidity accumulated for all ecosystems in a grid cell. The purpose of using the accumulated excess is to avoid the undue influence of a specific ecosystem (percentile of the cumulative critical load distribution - Barkman, 1997) and thus increase the robustness of the modelling results. In the RAINS optimisation routine, the fact that the accumulated excess function is convex also allows the use of the soft constraints and compensation mechanism described in Section 9.5.2.

9.5 Uncertainties and target setting

In addition to the consideration of uncertainties within the framework of the model itself, attempts were made to limit the effect of uncertainties on the model optimisation outcome by selecting an appropriate method of setting the optimisation targets. In practice, the potential influence of uncertainties was minimized by using ‘gap closure’ targets (relative improvements), by developing a compensation mechanism for targets, through the use of explicit model confidence intervals, and by excluding extreme situations.

Examining a range of scenario optimisations for different combinations of targets, rather than relying on one 'central' scenario alone, further enhanced the reliability of the resulting emission ceilings.

9.5.1 Use of relative 'gap closure' targets

The occurrence of and the reduction potential for ground-level ozone and acidification show distinct spatial differences across Europe. Furthermore, there is robust evidence that the presently available technical emission control measures will not be sufficient to meet the environmental long-term targets (the no-damage levels) everywhere in Europe within the next one or two decades without interfering with the 'business as usual' expectations on economic development and energy consumption. In such a situation the choice of an equitable environmental interim target becomes crucial for deriving a balanced emission control strategy. Two basic concepts for setting interim targets have been considered:

- Prioritising measures in highly polluted areas by imposing uniform absolute exposure limits over the entire area;
- Postulating equal relative improvements in relation to the situation in a base year (the gap closure concept). This approach, involving relative improvements, is less prone to model uncertainties because by focusing on the differences between scenario results, some of the potential biases that apply for absolute model results cancel out. Thus, such a gap closure concept provides a more reliable target-setting framework than one based on absolute limits.

However, these two different conceptual approaches imply fundamentally different spatial distributions of environmental benefits and emission abatement efforts over Europe. These differences were explored in close interactions with the negotiating bodies, and a combination of both principles eventually proved acceptable to all Parties.

9.5.2 Compensation mechanism for targets

Earlier analysis demonstrated that the optimal allocation of emission controls might be strongly influenced by the need to exactly meet specific environmental targets at a few single grid cells, while for the majority of grid cells the targets are usually over-achieved. The sensitivity of the optimisation results towards modifications of the environmental targets of these 'binding grids' was the subject of numerous discussions in the past. It was argued during the policy negotiations that the requirement to achieve stringent targets in isolated areas could possibly imply unbalanced high costs without yielding adequate benefits. This concern is even more pronounced when the targets are not related to absolute exposure levels, but to interim targets on the way towards the ultimate environmental objective. Alternative concepts, in which the environmental targets for single ecosystems are not allowed to drive the overall optimisation system to extreme solutions, are necessary to overcome this problem.

In order to limit the potential influence of small and perhaps atypical environmental receptor areas on optimised Europe-wide emission controls, and to increase the overall cost-effectiveness of strategies, a mechanism was developed to tolerate lower improvements at a few places without discarding the overall environmental ambition levels. This 'compensation mechanism' allows a (limited) violation of environmental targets at single grid cells or in single years as long as this excess is compensated by additional improvements in other years or at other grid cells within the same country. The compensation considers differences in the stock at risk over grid cells and puts more relative emphasis on densely populated areas or regions with large natural ecosystems. A weighting mechanism requires that excess exposure (AOT60, AOT40 or accumulated excess acidity/nitrogen) must be compensated on a population- or vegetation-adjusted basis, e.g., a small excess of AOT60 in a big city by larger improvements in less populated rural areas. The country balances ensure that for each country the exposure indices will be reduced by at least the percentage of the selected gap closure, or phrased differently, that the desired 'gap closure' is achieved for the country population/vegetation exposure indices rather than for individual grid cells.

In order to avoid a possible inequitable treatment of large and small countries implied by the compensation mechanism, a (uniform) maximum compensation potential was introduced. This means that environmental targets may only be violated up to a certain amount, which is independent of the country. Experiments showed that such a violation limit was best defined in terms of a uniform 'minimum' gap closure, compared to other relative or absolute measures.

This compensation mechanism was also examined in terms of its economic meaning (Forsund, 2000).

9.5.3 Explicit model confidence intervals

Earlier analysis also revealed that in certain situations the original definition of the 'gap' (the difference between present and absolute 'no-damage' levels) could push areas with comparatively low exposure to costly emission reductions, while less burden would be placed on more polluted regions. This occurs typically in areas where background concentrations resulting, e.g., from natural sources, constitute a large fraction of the total exposure. At such places, a target specified as a certain relative improvement requires higher reductions in anthropogenic emissions than in highly polluted regions, where the relative contribution of natural background is negligible.

It is important to recall that model uncertainties are, for a number of reasons, largest for just these low pollution levels. In order to maximize the robustness of results obtained from the currently available models and to prevent extremely low model results from influencing the actual strategy development, a 'model confidence interval' was introduced. The 'gap to be closed' by the optimisation is now defined as the difference between the current situation and this model confidence interval. In practice, the lower model confidence range was set for the AOT60 to 0.4 ppm.hours and for acidification for each grid cell to the accumulated excess deposition resulting from natural and hemispheric background plus five aeq/hectare.

9.5.4 Excluding extreme situations

In addition to the general gap closure targets, general exposure ceilings to be achieved throughout the modelling domain –as limits to the permitted violations of the gap closure targets - were also introduced. These uniform exposure ceilings proved to be practical tools to exert additional pressure for environmental improvements in the most polluted areas.

For ozone however, model results for five different meteorological years demonstrated that actual ozone levels do not only depend on the levels of precursor emissions, but also to a significant degree on the specific meteorological conditions. Emission control strategies addressing an extreme situation might therefore look rather different from strategies tailored towards the improvement of typical situations. For the purposes of strategy development, it was decided to exclude the 'most difficult' situations from the analysis, when considering the uniform ozone limit target. In practice, the strategy should be constructed in such a way that it would meet the absolute AOT targets in four out of five years. It is important to stress that the major motivation for this 'four out of five' principle in the context of strategy development is the concern to avoid undue reliance on model performance for extreme (and perhaps rare) situations.

9.6 Sensitivity analyses

Additional optimisation analyses were performed to explore the sensitivity of the optimisation results to changes in important input assumptions. The projected level and composition of energy use can be major determinants of the internationally optimised allocation of emission reductions, as can the agricultural policy assumed. The scenarios constructed to examine such factors included a 'post-Kyoto' (or low CO_2) scenario to give an overall indication of the possible impact of the Kyoto agreement, a 'high SO_2' scenario to reflect more combustion of sulphur-containing fuels than foreseen in the energy projection or less efficient SO_2 emission controls, and both 'low NH_3' and ' high NH_3' scenarios to investigate the possible effects of the uncertainties associated with the forecasts of livestock numbers and the efficiencies of emission control options.

In general, the results of these sensitivity analyses suggest that optimised emission reduction levels appear to be robust towards (limited) increases in projected activity rates, reduced emission control potentials and increased costs for emission controls. Lower activity rates, however, do generally result in lower emission levels and a relaxation of the most expensive emission controls. Cost savings for scenarios with low activity rates can be substantial.

9.7 A method for error propagation in RAINS

In 1999, the RAINS model was expanded to examine how errors (quantified uncertainties) in the input parameters propagate through the RAINS model calculations from economic activity to the protection of ecosystems (Suutari et al., 2001). While a methodology has been developed, it was found much more difficult to reliably quantify uncertainties on a solid basis than, e.g., mean values that are used in traditional deterministic analyses. Therefore, the quantification of the **uncertainties** themselves is considered as the **most uncertain** element in the **uncertainty analysis**. Indeed, many assumptions on the CV of input data to the different modules could be justified only in a very tentative manner and further work will be necessary to improve the understanding for quantifying uncertainties of input data.

This observation holds despite the methodological approach, which relies solely on the first and second moments of model parameters (i.e., the means, the variances and correlations) and, in contrast to many other approaches to uncertainty analysis, does not require assumptions about distributions, which are even more difficult to establish on a firm basis.

Furthermore, it was found most difficult to quantify (in several cases even the sign of) correlations between input parameters. As a consequence, only a very limited number of correlations could be considered, accepting that this limitation could have bearings on the conclusions of the analysis. The extent to which the variability of RAINS outputs regarding protected ecosystems is affected by the ‘uncertainty of uncertainties’ or the ‘uncertainty of expert opinion’ needs to be a central subject of future work.

The error propagation methodology developed for RAINS is only applicable to additive and multiplicative models. It cannot be used for non-linear models, e.g., for determining uncertainties of critical loads from input parameters, or more generally, for any process involving ranking/substituting of options.

It is pointed out in this paper at several places that quantitative uncertainty estimates can only be computed for specific model outputs. General notions like ‘the uncertainty of a given model’ do not appear particularly useful concepts. Different types of model output have different uncertainties, as demonstrated, e.g., in the case of sectoral and national total emissions. This also reinforces the basic concept that each model has its specific purpose for which it was constructed and for which the control of uncertainties is a critical issue. Using the same model for other endpoints might put the uncertainties in a completely different context and requires careful analysis of these implications.

9.7.1 Uncertainty of emission estimates

It was found that the uncertainties in calculations of emissions, which add up a large number of multiplicative operations for individual sources, are strongly determined by the potential for error compensation. This potential is larger - and therefore the uncertainties are smaller – if more elements of similar sizes are included and if there is no (emission) source that makes a dominating contribution.

Therefore, in general, RAINS model estimates of sectoral emissions are more uncertain than national total emissions. The error compensation leads to the situation that in many countries levels of national SO_2 emissions turn out to be more uncertain than those of NO_x and even NH_3, despite uncertainties in many of the input parameters for NO_x and NH_3 calculations being larger than those for SO_2.

This finding has an implication on the optimal design of emission inventories, suggesting that more resolved emission inventories should be associated with less overall uncertainties. However, the potential for such improvements is limited by the associated need for additional information at the more resolved level with equal quality. Simple disaggregation of sectors without additional genuine information would just introduce strongly correlated terms in the analysis, which in turn will not influence the uncertainties of the overall estimate.

Real improvements in emission inventories are inextricably linked to the availability of additional information and deeper insight into the correlations of parameters.

9.7.2 Uncertainties of deposition estimates

For practical reasons the analysis could not explore the full range of potential factors that contribute to the uncertainties in the estimates of atmospheric dispersion of pollutants. Further insights could be gained through additional analysis, e.g., with the EMEP dispersion model.

The analysis, which focused on the inter-annual meteorological variability, shows that error compensation is also an important mechanism for deposition estimates, with direct impacts on the uncertainties of results. Deposition estimates are more uncertain if deposition at a given site is dominated by the emissions of single source (region), e.g., at the Kola Peninsula and in Romania. Also an uneven distribution of emissions within a country (e.g., if there is only one large power station making a dominant contribution to national emissions) leads within the country to larger uncertainties in the deposition field due to the inter-annual meteorological variability.

In general, the combined uncertainties of emission estimates and of the inter-annual meteorological variability leads to similar uncertainties of sulphur and reduced nitrogen (NH_3) deposition fields, while for oxidized nitrogen uncertainties turn out to be slightly lower.

The analysis reveals an interesting aspect showing lower uncertainty in the field of total nitrogen deposition than for the two individual components. This is caused by a negative correlation between NO_x and NH_3 deposition, which was identified from the EMEP model calculations for areas close to emission sources as a consequence of meteorology and short-term chemical reactions. Sunny and warm weather increases the conversion rate from NO to NO_3^-, which shortens the travel distance of nitrogen oxides and increases local deposition. For ammonia, larger rates of local (wet) deposition occur in wet weather conditions.

9.7.3 Uncertainties of estimates on ecosystems protection

Suutari et al. (2001) developed and implemented a method to analyse the uncertainties in the protection of ecosystems due to the uncertainties of critical loads on a European scale. To get the most out of such an analysis, good quality data characterizing the uncertainties is needed, i.e., the analysis should not be questioned due to the "uncertainties in the uncertainties". An uncertainty analysis such as that of Suutari et al. not only provides information on the confidence levels which can be assigned to IAM results, but can also help to identify those parameters for which better knowledge can most improve the accuracy and precision of the overall results.

For estimates of ecosystems protection, the spread of critical loads within a grid cell appears to have a stronger impact on the resulting uncertainties and possible error compensation (as long as perfect correlation is assumed). This means that in cases where countries report only few critical loads for grid cells or where these critical loads are in a similar range, estimates of ecosystems protection are rather uncertain since, e.g., a small change in these data or in deposition might change the protection status for many ecosystems.

Furthermore, it is important to point out that the traditional deterministic calculations of the RAINS model represent the median of the probability distribution and thereby assume a 50 percent probability of the achievement of the environmental targets. It is clear from the calculations that there is a significant uncertainty interval around the median and, depending on the level of confidence one puts into the calculations, the achievement of the original policy target appears in a different light.

As a conclusion, setting of interim or long-term environmental policy targets should not only address the desired level of protection but at the same time also consider the certainty with which this level should be achieved. The uncertainty range is considerable, and it needs to be explored how different confidence levels will influence the economic efforts that are needed to attain them.

9.8 Workshop on Treatment of Uncertainties in Integrated Assessment Models

In January 2002, well before the next round of policy development for air pollution control in Europe, IIASA organized a workshop to review and discuss possible approaches for the treatment of uncertainties in integrated assessment modelling. The workshop discussed the needs and wishes of decision makers with regard to uncertainties and robustness, reviewed how uncertainties were addressed in past work in the context of air pollution and climate assessments, and explored practical approaches for refined treatment of uncertainties for the next round of assessments.

9.8.1 Some basic points

The workshop identified six steps towards uncertainty management in an interactive learning process, with good interaction between all actors involved from science to policy:

- Denial of uncertainties;
- Acknowledgement of uncertainties;

- Specification of types of uncertainty;
- Quantitative assessment of uncertainties;
- Specification of policy relevance;
- Uncertainty management.

The workshop also identified different types of uncertainty. Some can be handled with statistical and other methods, others cannot be dealt with in the same manner. Many uncertainties can be reduced through further research. They result from incomplete scientific understanding of the various processes (for instance, manifested in the different predictions from different modelling systems, different meteorological models or parameterisations, different chemical mechanisms, or uncertainties in the emissions inventories). Some uncertainties are inherent and cannot be reduced. They are caused by processes that operate on space/time scales that cannot be captured by the models. A good process can help (and has helped) to deal with uncertainty: transparency, participation, and consensus building around scenarios.

9.8.2 The role of uncertainties in decision making

For the decision-making process, good communication of uncertainties and full transparency were recognized as crucial. The process involves three distinct groups that can in turn be split into sub-groups:

- Scientists (applied and basic);
- Integrated assessment modellers;
- Decision-makers (politicians and their representatives in negotiations), stakeholders and the public.

It was recognized as important to follow a systematic approach to uncertainties in order to gain confidence. Such an approach should differentiate between the reducible and the irreducible uncertainties. For the most significant sources of reducible uncertainties, it should determine by how much further scientific effort could increase the robustness of the models. For irreducible uncertainties, the model has to make assumptions. These should be made explicit and, where they significantly influence the model outcome, alternative scenarios should be explored.

Simple parameters can help to present results to policy makers and the public. For example, instead of presenting expected exceedances in absolute or deterministic figures, maps can show the probability of exceedance.

Policy makers are aware of uncertainty. They are interested in the sources of uncertainty and whether/how they can be reduced.

Decision-makers are looking for a rational basis for decisions, but, in the end, various driving forces often dominate specific decisions. The reliance on model results will be higher (independent of uncertainties) if model results fit the political driving forces. There is a risk that uncertainty is misused as an argument for delay, when there are opposing scientific and/or political positions.

Policy makers, in contrast to scientists, are not interested in the detailed statistics about uncertainties. They are interested in robust strategies. Robustness implies that strategies (control needs and priorities between countries, sectors, pollutants) do not significantly change due to changes in the uncertain model elements. Robust strategies should avoid regret investments (no-regret approach) and/or the risk of serious damage (precautionary approach). As part of such a strategy, the timing of measures may be a risk management tool. The choice of the time horizon will also influence robustness.

9.8.3 Uncertainties in the model chain of integrated assessment modelling

The meeting concluded that "good science includes a full uncertainty assessment. Such assessment is also necessary for the communication between scientists working within the Convention framework and other scientists. Scientific debate about uncertainty should lead to consensus on which uncertainties are the most important. It should flag any fundamental flaws and highlight specific points of disagreement."

Integrated assessment modelling should construct the model in a way to avoid the results being driven by the most uncertain elements. If this is impossible, it should present different scenarios to illustrate the importance of specific uncertainties.

In the discussion with decision makers some questions about the usefulness of a quantitative uncertainty analysis for policy analysis were raised:

- Are Parties ready to put increased effort into providing and, subsequently, agreeing upon the data needed for such an analysis?
- Would Parties be prepared to follow abatement strategies derived with such a method, i.e. to pay more for strategies that yield the same environmental improvements but with a higher probability of attainment?

Practical answers to these questions need to be provided before the substantial investment of implementing this approach for the RAINS calculations on a routine basis could be made.

9.8.4 Conclusions

A full and systematic assessment of the role of individual model and data uncertainties in an integrated assessment model such as RAINS is a complex and time-consuming task. A practical, alternative approach to uncertainty treatment in the context of policy-oriented applications has been adopted, however, in which several precautionary measures are taken to limit the influence of the most uncertain model elements on the optimisation results. The environmental targets were selected in such a way that the confidence ranges in model performance are taken into account. Furthermore, extreme values in critical loads estimates (the low percentiles) were disregarded when setting the environmental targets, and the revised cost-curve routine excludes measures with questionable cost-effectiveness (e.g., retrofits of already controlled plants, etc.).

Sensitivity analyses have been performed, showing that the optimised emission ceilings are generally robust against higher activity rates, but would decline for scenarios with lower rates of economic activity. This is of particular importance for analyses based on a pre-Kyoto scenario, which is incompatible with the existing agreements - although yet to be ratified - for meeting the Kyoto targets on controlling the emissions of greenhouse gases.

It remains to the user of the analysis to interpret the optimisation results in the light of the known but still unquantified uncertainties. In order to minimize the potential flaw from theoretically possible model biases, it is clear that more trust should be put in the relative changes (e.g., percent change compared to the base year) than in absolute numbers on resulting emission levels.

9.9 References

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